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DEMOGRAPHY OF DESERT MULE DEER IN SOUTHEASTERN CALIFORNIA

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Desert mule deer, *Odocoileus hemionus eremicus*, occur at low densities in the Sonoran Desert of southeastern California and consequently are difficult to monitor using standard wildlife techniques. We used radiocollared deer, remote photography at wildlife water developments (i.e., catchments), and mark-recapture techniques to estimate population abundance and sex and age ratios. Abundance estimates for 1999-2004 ranged from 40 to 106 deer, resulting in density estimates of 0.05-0.13 deer/km². Ranges in herd composition were 41-74% (females), 6-31% (males), and 6-34% (young). There was a positive correlation ($R = 0.73$, $P = 0.051$) between abundance estimates and number of deer photographed/catchment-day, and that relationship may be useful as an index of abundance in the absence of marked deer for mark-recapture methods. Because of the variable nature of desert wildlife populations, implementing

strategies that recognize that variability and conserving the habitat that allow populations to fluctuate naturally will be necessary for long-term conservation.

INTRODUCTION

Desert mule deer, *Odocoileus hemionus eremicus*, occur in the Sonoran Desert of southeastern California (Bowyer and Bleich 1984) at low densities (Thompson and Bleich 1993) and in a scattered distribution (Celentano and Garcia¹ 1984). Efforts to quantify deer population parameters in that area have been difficult because of low densities and low detection probabilities (Thompson and Bleich 1993). Celentano and Garcia¹ (1984) estimated sex and age ratios using aerial and ground telemetry, and Thompson and Bleich (1993) evaluated methods (aerial and ground surveys, hunter interviews) to estimate deer herd composition. The methods of Thompson and Bleich (1993) produced proportions of adult males, adult females, and young-of-the-year, but not estimates of abundance. Annual deer harvest records for this region may also provide useful information about the general trend of this deer population (Marshall et al. 2002). However, further efforts to quantify this deer herd should move beyond indices of abundance and trend, and toward more rigorous population estimates.

Recent efforts to quantify the demographics of this deer population have used remote photography (Kucera and Barrett 1993) of deer visiting wildlife water developments (i.e., catchments) to estimate population parameters. Remote photography, in combination with marked animals, allows for use of mark-recapture methods to estimate population abundance (Martorello et al. 2001, Heilbrun et al. 2003). Our objectives were to develop remote photography methods for population studies of desert ungulates and to use those methods to estimate abundance and composition of mule deer in a region of the Sonoran Desert in southeastern California.

STUDY AREA

We monitored the deer population from 1999 to 2004 in a region of the Lower Colorado River subdivision of the Sonoran Desert (Brown 1994) in eastern Imperial County near the East Chocolate and Cargo Muchacho Mountains (1,681 km², Fig. 1). Temperatures ranged from 0°C in winter to >45°C in summer. Annual rainfall was low, but highly variable (mean = 74 mm, range = 4-216 mm; 1914-2003, Imperial Irrigation District, unpublished data). The study area contained three major landforms: mountain, piedmont, and flat (Andrew et al. 1999). Vegetation was most dense in xeroriparian zones around washes (Marshall et al. 2005) and consisted largely of desert ironwood, *Olneya tesota*, creosote bush, *Larrea tridentata*, and palo verde, *Cercidium floridum* (Andrew² 1994). Outside of xeroriparian zones, creosote bush, and cholla and prickly

¹Celentano, R. R. and J. R. Garcia. 1984. The burro deer herd management plan. California Department of Fish and Game, Sacramento, California, USA.

²Andrew, N. G. 1994. Demography and habitat use by desert-dwelling mountain sheep in the East Chocolate Mountains, Imperial County, California. Thesis, University of Rhode Island, Kingston, Rhode Island, USA.

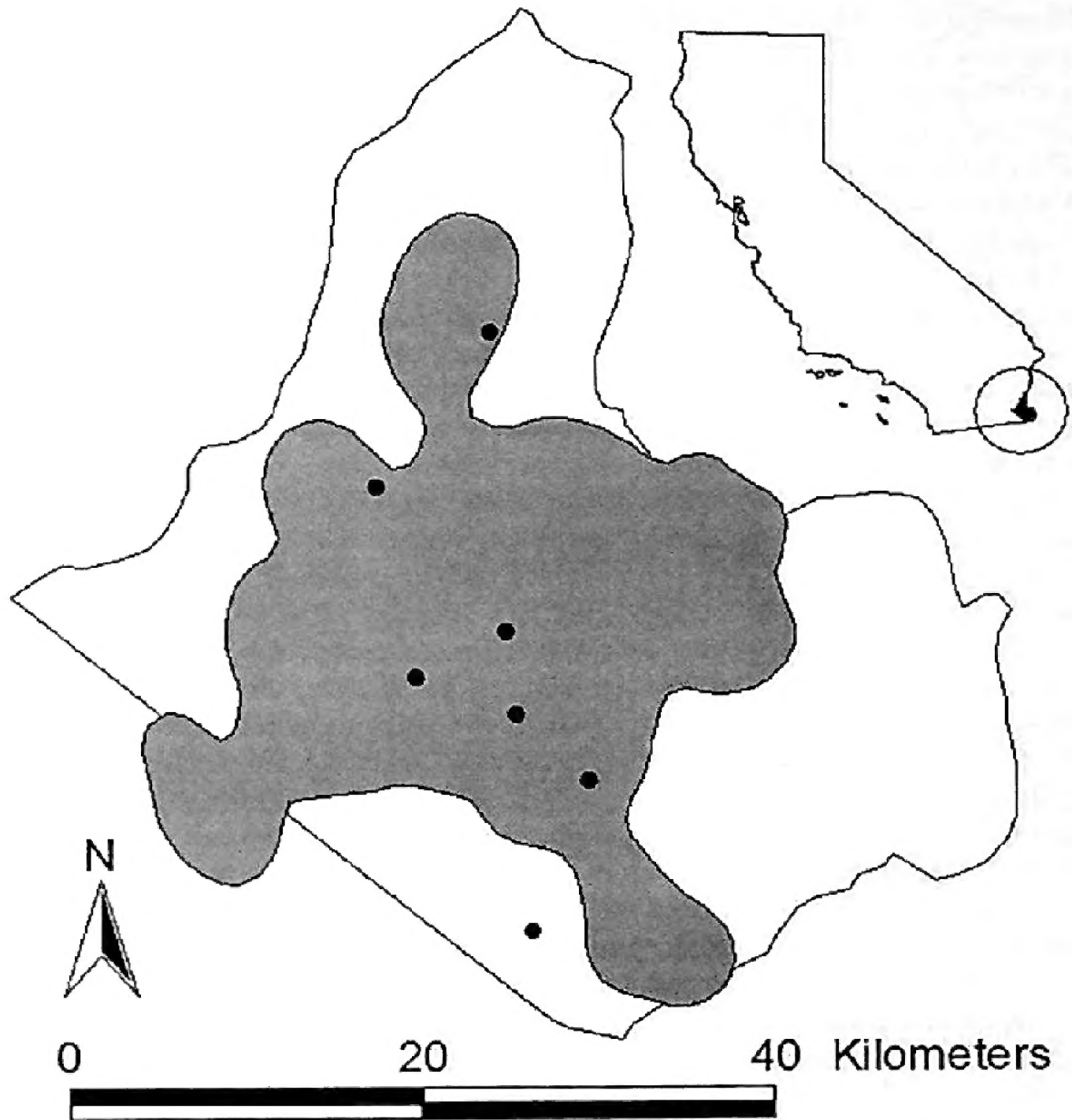


Fig.1. East Chocolate-Cargo Muchacho study area showing locations of the seven catchments at which radiocollared deer were photographed (circles), and the population range polygon for estimating density (grey area), Imperial County, California.

pear, *Opuntia* spp., were common (Andrew² 1994). Plant nomenclature follows Munz (1974). Other large- and medium-sized herbivores in the area were bighorn sheep, *Ovis canadensis*, feral ass, *Equus asinus*, black-tailed jackrabbit, *Lepus californicus*, desert cottontail, *Sylvilagus audubonii*, and desert tortoise, *Gopherus agassizii*. Potential predators of mule deer included mountain lion, *Puma concolor*, coyote, *Canis latrans*, bobcat, *Felis rufus*, and golden eagle, *Aquila chrysaetos* (Andrew et al. 1997). Hunting season for mule deer occurred in October and November. Since 1990, approximately

15-60 deer have been harvested each year from the hunting zone that contains our study area (Marshall et al. 2002).

There were several sources of free water available to deer. The Colorado River and the All-American Canal were at the eastern and southeastern boundaries of the study area. Seventeen catchments occurred in the study area, at an average density of 1/35 km²; these provided water in all seasons. Natural rock basins and springs also occurred, but many were dry for part of the year. We defined seasons according to patterns of rainfall and temperature (Marshall et al. 2005). Generally, winter (January-March) was cool-rainy, spring (April-June) was hot-dry, summer (July-September) was hot-rainy, and autumn (October-December) was cool-dry. Radio-collared mule deer moved to catchments (within approximately 5 km) during spring, where they remained until the first rains of the hot-rainy season (Marshall et al., unpublished data). Timing of movements varied between years, and depended on forage conditions the previous winter and timing of the first summer precipitation.

METHODS

Capture and handling of mule deer occurred in accordance with protocols approved by the California Department of Fish and Game (Jessup et al. 1986). We captured deer with a net-gun (Krausman et al. 1985) fired from a helicopter, with chase times limited to ≤ 5 minutes. Upon capture, we blind-folded and hobbled deer to facilitate handling and to protect the animal (Jessup et al. 1986). We fitted each deer with a VHF radio collar with 6-hour delay mortality signal (Telonics MOD-500, Mesa, Arizona, USA) prior to release. To establish when deer moved to catchments, we located radio-collared deer approximately weekly from a fixed-wing aircraft.

Once deer moved to the catchments, we used infrared remote triggers (TrailMaster 1500; Goodson and Associates, Inc., Lenexa, Kansas, USA) and automatic film cameras to photograph them at the catchments. We placed the emitter and receiver on either side of the access point for a catchment. The triggers were connected to a camera via a cable, and the camera was positioned to facilitate identification of deer at the catchment as adult male, adult female, or young-of-the-year (10-12 months of age at the time of photographic sampling). This distance was typically 3 m, but varied with the surrounding terrain and vegetation. Sensitivity of the infrared units was set low (setting = 5) to trigger a photograph only if a large object broke the infrared beam, and the timer was set to take photographs > 20 minutes apart. Photographic sampling occurred at 2-7 catchments (Fig. 1), depending on the number visited by radio-collared deer, and continued until deer moved away from the catchments in early summer.

We used radio-collared deer and two-sample Lincoln-Peterson mark-recapture methods to estimate population abundance in the study area. We estimated abundance from observations of collared and uncollared deer at visited catchments. We estimated abundance with Chapman's (1951) bias-adjusted mark-recapture estimator:

$$\tilde{N} = [(n_1 + 1)(n_2 + 1) / (m_2 + 1)] - 1,$$

where \tilde{N} was the population estimate, n_1 was the number of collared deer in the population, n_2 was the number of deer photographed, and m_2 was the number of collared deer photographed. We determined n_1 from the number of collared deer observed near catchments after they had moved to them in the spring. After 2001, we estimated the number of collared deer in the study area because of transmitter failures. To determine a number of marked deer during 2002, 2003, and 2004, we calculated annual average survival rate for transmitting deer using the Kaplan-Meier estimator (Pollock et al. 1989). From survival rate, we estimated the number of nontransmitting deer that would have been alive to be seen during photographic sampling (Appendix 1). We assumed that nontransmitting deer had the same survival rates as transmitting deer, that transmitter failure and not emigration was occurring, and that transmitter failure was independent of the fate of the animal.

Values for n_2 and m_2 came from direct counts of deer in photographs; we did not identify individual deer (i.e., we sampled with replacement). As a consequence, $m_2 > n_1$, which produced a negative value for the estimate of $\text{var}(\tilde{N})$, based on Seber's (1970) formula for variance. Therefore, we estimated 95% confidence intervals using bootstrap methods (Manly 1997). We selected, at random and with replacement, a number of week-long sampling periods and their associated deer counts equal to the total number of sampling periods for one year. From each random sample of periods, we calculated a population estimate. We repeated this procedure 1,000 times to produce a bootstrapped population estimate (\tilde{N}_{bs} , the average of the 1,000 estimates), and 95% confidence intervals. We calculated density by dividing \tilde{N}_{bs} by the area occupied by the population. We calculated this area by estimating a home range polygon around all locations for all collared deer with a 95% adaptive-kernel home range estimator (Worton 1989).

Males were identified by presence of antlers. Females were distinguished from young by face and body proportions; young had smaller, thinner bodies, and their faces were narrower and smaller relative to the size of ears. We combined all young into a single category. In cases where distinguishing characteristics of deer could not be identified in photographs, we categorized those deer as undetermined. We calculated annual proportion of deer in each age and sex category using counts of deer from photographs (excluding deer classified as undetermined), and we calculated ratios of males:100 females and young:100 females. Because we were unable to identify individual deer when classifying them, we could not determine a number of independent trials to use when estimating variance based on the binomial distribution. Consequently, we used bootstrap methods similar to those described earlier for abundance estimation to calculate 95% confidence intervals for age and sex ratios.

To develop an index of abundance that would not rely upon a sample of collared deer in the population, we compared population abundance estimates to total observations of photographed deer/catchment-day. We tested for a positive association by using Pearson's product-moment correlation.

RESULTS

Over 5 years, we captured 47 female and 7 male deer. The population home-range size was 817 km² (Fig. 1), based on 4,013 deer locations. Estimated abundance of deer was variable among years, ranging from 40 deer (95% CI: 36-46) in 2001 to 106 deer (95% CI: 77-138) in 2002 (Table 1). Average abundance over the 6 years was 79 deer, and the coefficient of variation was 32%. Based on the estimated population range size, density was 0.05-0.13 deer/km² (Table 1). There was evidence for a positive association between \tilde{N}_{bs} and number of observed deer photographed/catchment-day ($R = 0.73$, 1-tailed $P = 0.051$). Deer herd composition was also variable. Males: 100 females ranged from 9 to 61 in 1999 and 2004, respectively (Table 2). Young: 100 females ranged from 10 in 2001 to 85 in 2004 (Table 2). Ranges in percent of females, males, and young were 41-74, 6-31, and 6-34, respectively (Table 2).

Table 1. Population abundance (\tilde{N}_{bs}) for desert mule deer in the East Chocolate-Cargo Muchacho area of Imperial County, California 1999-2004.

Year	\tilde{N}_{bs}	95% LCL	95% UCL	Density (deer/km ²)	Deer/catchment-day
1999	56	38	68	0.07	15.6
2000	90	76	108	0.11	11.6
2001	40	36	46	0.05	6.2
2002	106	77	138	0.13	27.4
2003	87	70	111	0.11	26.4
2004	95	70	134	0.12	17.5

Table 2. Bootstrap-estimated sex and age composition of mule deer in the East Chocolate-Cargo Muchacho Area, Imperial County, California, 1999-2004.

Year	Catchment -days	No. deer classified	Males:	Young:	% Females (95% CIs)	% Males (95% CIs)	% Young (95% CIs)
			100 females (95% CIs)	100 females (95% CIs)			
1999	20	312	9 (5-11)	28 (19-31)	74 (71-80)	6 (4-8)	20 (15-22)
2000	66	766	33 (15-59)	17 (10-27)	67 (54-78)	22 (12-33)	11 (7-16)
2001	95	589	55 (30-85)	10 (2-24)	62 (50-72)	31 (21-43)	6 (1-13)
2002	39	1070	38 (16-69)	71 (55-93)	48 (40-55)	18 (9-29)	34 (28-41)
2003	23	608	40 (21-68)	43 (32-54)	55 (48-60)	21 (13-34)	24 (16-30)
2004	28	491	61 (35-102)	85 (56-113)	41 (33-48)	24 (16-36)	34 (25-42)

DISCUSSION

Our findings confirm the conclusions of Celentano and Garcia¹ (1984) and Thompson and Bleich (1993) that this is a low-density mule deer population. Comparisons to other parts of desert mule deer range support this conclusion. For example, density of mule

deer in the Belmont Mountains, Arizona, was 0.5/km² (Albert and Krausman 1993). Densities of desert mule deer ranged from 0.7 to 4.2/km² in the Chihuahuan Desert, Durango, Mexico (Sánchez-Rojas and Gallina 2000). Densities of desert mule deer in Buenos Aires National Wildlife Refuge, Arizona, ranged seasonally between 0.9 and 2.5 deer/km² (Koenen et al. 2002). Martinez-Muñoz et al. (2003), however, reported a desert mule deer density as high as 13.3/km² in Coahuila, Mexico. Historical information on the number of deer in our study area is scarce. McLean (1940) reported a herd size of 920 deer that occupied an area of approximately 11,900 km² (0.08 deer/km²) covering eastern Imperial and Riverside counties next to the Colorado River. Longhurst et al. (1952) reported that the deer population in a similar region contained 2,000 individuals (0.11 deer/km²). These historical numbers fell within our range of densities over the 6 years of our study (0.05-0.12 deer/km²).

Low density of deer in this region is likely the result of low rainfall, and its consequences for production of deer forage (Marshal et al. 2005). However, a low average rainfall masks the highly unpredictable and dynamic nature of this arid system. In arid environments, variation in rainfall leads to variation in plant biomass, and each is rarely near the long-term average (Noy-Meir 1973). Forage conditions in this arid region of California match this scenario (Marshal et al. 2005). Deer populations likely responded demographically in a manner that produced a similarly variable trend in abundance. As a result, the range in deer abundance estimates appeared wide and the coefficient of variation for abundance large. This would not be unexpected in a plant-herbivore system driven by stochastic rainfall events (Caughley 1987). Such fluctuations may have contributed to the relatively high and low population estimates observed during this study (Table 1). Movements of deer into or out of the study area probably contributed little to this variation; evidence from the radio-collared deer indicated little movement of deer to areas neighboring the study area (Marshal et al., unpublished data).

Using our methods, we were unable to estimate seasonal changes in deer density. Such changes could occur because of movements of deer into or out of the study area, or when the same number of deer use a larger or smaller area. Some studies of desert mule deer have reported fairly large changes in density across seasons. Densities estimated by Koenen et al. (2002) show a marked change between summer (0.9 deer/km²) and winter (2.5 deer/km²) over a single year. Desert mule deer densities in Durango, Mexico varied considerably across seasons, but not consistently by season. Sánchez-Rojas and Gallina (2000) estimated June densities of 3.6, 0.7, and 2.1 deer/km² during their 3-year study. Because arid environments are highly variable (Noy-Meir 1973), describing seasonal variation in deer density may be less useful for understanding population ecology of this species than describing density responses to specific environmental conditions. In our study area, density may increase during hot, dry conditions because of a decrease in the area used as animals moved toward water sources and, consequently, ranged less widely, rather than because of a change in the number of animals in the area. We considered this possibility and estimated a 95% adaptive-kernel home range polygon for locations during spring. The difference between home range sizes was not substantial (annual: 817 km²; spring only [1,031

locations]: 811 km²).

Age ratios involving young-of-the-year or yearlings:100 females are frequently used as indices of reproduction or juvenile recruitment, respectively. At the time that photographic sampling occurred, young were nearing their first birthday. A measure of the proportion of these "near-yearlings" would provide a reasonable estimate of the rate at which young animals are becoming adults, assuming no variation in the rest of the population (McCullough 1994). Biologists should exercise caution when using age ratios as a measure of recruitment to a population. Using simulations to investigate the use of age ratios in evaluating the dynamics of wildlife populations, Caughley (1974) reported that populations exhibiting the same rates of increase could have very different age ratios, depending on what was occurring in the adult part of the population. Further, populations with the same age ratios could be exhibiting very different dynamics. For example, a population may have an increasing proportion of young because of an increase in the number of young animals, as might occur in an increasing population. But such a ratio also may be due to a large mortality event among adults that results in a decreasing population. While the use of age ratios may provide some information about population trend, they are unreliable in the absence of supporting data, such as rate of change estimates (Caughley 1974, McCullough 1994). In the absence of such data, use of age ratios requires the assumption of low variability in adult mortality (McCullough 1994). There is evidence that, for ungulate populations, most mortality occurs in the few months following birth (White and Bartmann 1998, Gaillard et al. 2000, Bleich et al. in press); in our study area, early fawn mortality also likely fluctuates considerably, and may do so in response to rainfall and forage conditions.

Despite the shortcomings associated with use of age ratios, our study had the support of estimates of abundance, and from abundance, estimates of rate of change were possible. Further, additional information on the survival of adults is available from telemetry data that may assist in interpreting the dynamics of the adult portion of the population. However, in the absence of survival data and if monitoring of this population occurs entirely by remote photography (i.e., without the assistance of radio collars or marked animals), use of age ratios alone may not allow for an adequate understanding of the dynamics of this deer population.

One alternative to age ratios may be to rely upon predictable relationships between rainfall and deer abundance to understand current deer dynamics and to predict future dynamics. Indeed, Marshal et al. (2002) used deer harvest as an index to deer abundance and reported a positive linear association between harvest in any year and rainfall the previous year. That relationship included a large amount of variation, so efforts to predict deer abundance may require relationships between more closely related variables, such as between browse biomass and deer abundance rather than between rainfall and reported harvest. Another alternative may take advantage of the positive correlation between estimated deer abundance and number of deer observed in photographs/catchment-day. A longer-term study with a larger sample size would be necessary to produce a relationship that could predict deer abundance simply from photograph counts. Nonetheless, it does suggest a means to evaluate density and rate

of change of this deer population without the need for marked animals or the use of age ratios to evaluate dynamics.

The two-sample Lincoln-Peterson method has three assumptions: the population is closed to births, deaths, immigration, or emigration during sampling; all animals have an equal probability of capture or sighting in each sample; and marks are not lost, overlooked, or misinterpreted by the investigator (Otis et al. 1978). This population likely met the assumption of closure during photographic sampling. Observations from telemetry flights confirmed that collared deer moved little during spring and stayed relatively close to water sources (Marshall et al. unpublished data). During that time of limited resources, it was unlikely that animals would choose to leave a familiar area. For similar reasons, it was probably unlikely for new immigrants to enter the study area. Further, births did not generally occur in this area until after rains in summer, after which deer stopped visiting catchments and photographic sampling ended. Deaths among uncollared deer probably occurred over the sampling period; indeed, we lost some radiocollared deer during spring. But because of the short photographic sampling period, losses were probably minimal, and so the effect on bias was minimal.

Probability of detection in a photograph was a function of how frequently animals visited catchments to drink. We had no reason to expect a difference in frequency of drinking between collared and uncollared deer. However, a violation of this assumption may have resulted in a population estimate for 2001 that appeared relatively low and precise, compared to the other estimates during the study (Table 1). If, by chance, the number of marked deer in photographs was greater than would be expected from their proportion in the population, or if the number of unmarked deer was less than would be expected, population estimates would be biased low, but would also have relatively narrow confidence intervals, as observed for the 2001 estimate.

There may have been sex differences in drinking frequency that could influence sex ratios calculated from these data. Females in the Belmont and Picacho Mountains, Arizona, visited catchments to drink on average once/day, whereas males visited catchments once/1–4 days (Hervet and Krasuman 1986). Hazam and Krausman (1988), however, reported a drinking frequency of 1.1 times/day for males in the Picacho Mountains. If males did visit catchments less frequently than females, the result would be an underestimate of their proportion in the population.

The reliability of our herd composition estimates depended on the accuracy with which we identified adult females from young females. Factors that affect this ability include head or body position of deer when photographed (identifying features obscured), distance from the camera (further deer appear smaller and thus younger), and whether a young deer is standing next to an adult (identification by size comparison). Forage conditions may also affect the accuracy of identification, because in years with abundant forage, young grow more rapidly and by their first birthday appear to be similar in size to adults. When we were unsure about the classification of a deer, we categorized it as undetermined and removed it from subsequent analysis. Nonetheless, there exists the possibility of misclassification, which was a weakness of this method. For the cost and logistics, however, more accurate methods were not available.

An assumption that potentially was violated was that of no loss of marks. After

2001, transmitters began to fail, and we approximated the number of marked animals in the population. Our method involved estimating number of nontransmitting deer that survived to be photographed and was based on the assumptions that nontransmitting deer had the same survival rates as transmitting deer, that transmitter failure and not emigration was occurring, and that transmitter failure was independent of the fate of the animal. Because we could not monitor nontransmitting deer, we could not evaluate these assumptions. Nevertheless, there is some support for our method of estimating number of marked deer in the positive correlation between number of deer observations/catchment-day, which was not based on number of collared deer, and estimates of population abundance, which were. Overall, we believe these methods were useful for monitoring a low-density population of cryptic animals, and has allowed a level of statistical rigor that had not before been applied to this deer population.

Although it is clear that deer populations in California's Sonoran Desert are at low density, our results more importantly have demonstrated the highly variable nature of a population occupying an arid environment. Management of such populations is difficult, particularly where harvest is a management objective. Efforts to reduce variability and increase predictability in arid system are not likely to succeed, because they require management of factors outside the control of wildlife biologists (e.g., rainfall and forage availability). A more likely possibility is to implement a management strategy that recognizes the inherent variability of wildlife populations in arid environments and to conserve the natural systems (i.e., habitat) that allow those populations to fluctuate.

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Appendix 1. Information used to calculate population estimates in East Chocolate-Cargo Muchacho Area, California, 1999-2004.

Year	Catchments	Catchment -days	No. nontransmitting deer	Survival rate estimate	n_1^a	n_2^b	m_2^c
1999	2	20	0	-	14	312	79
2000	4	66	0	-	23	776	201
2001	4	95	0	-	21	587	314
2002	4	39	3	0.962	21 ^d	1070	221
2003	7	23	7	0.958	21 ^d	608	154
2004	4	28	14	0.788	20 ^d	491	108

^a No. collared deer in the population

^b No. deer observed in photographs

^c No. collared deer in photographs

^d No. marked deer in population estimated from number of nontransmitting deer and survival rate

OFF-HIGHWAY VEHICLE IMPACT ON THE FLAT-TAILED HORNED LIZARD, *PHRYNOSOMA MCALLII*, IN THE COLORADO DESERT OF SOUTHERN CALIFORNIA

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A randomly selected paired plot design, with dichotomous levels (high impact vs. low impact) of off-highway vehicle activity, was used to measure the effect of this activity on *Phrynosoma mcallii* (flat-tailed horned lizard). We measured lizard density, lizard mass, and food sources (ant mounds) on 4-ha square paired plots that were separated by 300 m. We selected paired plots from two study areas separated by 25 km in Imperial County, California. Low impact plots in the first study area contained greater lizard densities and mean lizard mass with suggestive statistical significance. High impact plots in the second study area contained greater lizard densities, but lizard mass remained greater on low impact plots. When analyzing the combined data from both study areas, only mean lizard mass was greater on low impact plots. For the first, second, and combined study areas, low impact plots contained greater ant mound densities (primary food source). We observed qualitative differences between the two study areas (e.g., recreational vs. non-recreational off-highway vehicle activity). Both recreational and non-recreational off-highway vehicle activity may have negatively impacted the lizard; we recommended increasing protection.

* Deceased.

INTRODUCTION

Human impacts on desert species concern land managers, conservation organizations, and private citizens, particularly in Arizona, California, and Mexico, where human activity has increased dramatically over the past several decades. One affected low desert species, *Phrynosoma mcallii*, (flat-tailed horned lizard, Phrynosomatidae), was twice considered for threatened status under the Endangered Species Act (USFWS 1997, USFWS 2003). In both 1997 and 2003, the U.S. Fish and Wildlife Service (USFWS) withdrew the proposal to list the species due to insufficient scientific evidence to warrant the listing (USFWS 2003). In an effort to minimize human threats, state and federal agencies formulated the flat-tailed horned lizard (FTHL) Rangewide Management Strategy in order to designate management areas for the lizard (Foreman 1997¹). However, substantial human threats occur within these management areas to *P. mcallii* that included habitat encroachment from agriculture, housing, roads, utilities, natural resource extraction, and impacts from off-highway vehicle (OHV) activity (Turner and Medica 1982, Foreman 1997¹, FTHLICC 2003²). Both recreational and non-recreational OHV impact was of specific interest to our study.

Studies on the effect of OHV impacts on arid ecosystems are extensive, and the 1997 and 2003 FTHL Rangewide Management Strategies review much of this literature (Foreman 1997¹, FTHLICC 2003² see also Webb and Wilshire 1983, Lovich and Bainbridge 1999). According to these reports, OHV impact may result in compacted soils, reduced vegetation cover, reduced diversity and biomass of reptiles, and direct mortality of desert species. Additionally, a few studies have examined the effect of OHV impact on *P. mcallii*. One study examined the effect of OHV impact on behavior and movement (Nicolai and Lovich 2000), which suggested that *P. mcallii* movement was reduced during an off-road racing event. However, Nicolai and Lovich (2000) did not examine the effect of OHV impact on *P. mcallii* density, which is of particular importance to land managers. In a report prepared for the California State Parks Off-highway Motor Vehicle Division, Setser (2001³) compared the occurrence of *P. mcallii* in areas impacted by OHV impact to randomly selected areas. This report suggested that *P. mcallii* occurred less frequently in areas impacted by OHV activity. Additionally, the Bureau of Land Management (BLM) annually monitors *P. mcallii*. According to Wright and

¹ Foreman, L. D. 1997. Flat-tailed Horned Lizard Rangewide Management Strategy. Report of Interagency Working-group, Bureau of Land Management, Riverside, CA. [Online, URL: <http://arizonaes.fws.gov/flat.htm>]

² Flat-Tailed Horned Lizard Interagency Coordinating Committee (FTHLICC). 2003. Flat-tailed Horned Lizard Rangewide Management Strategy, 2003 revision. [Online, URL: <http://arizonaes.fws.gov/flat.htm>]

³ Setser, K. 2001. Final Report: Scientific Study of Flat-tailed Horned Lizards, *Phrynosoma mcallii*, at Ocotillo Wells SVRA 2000 Field Season. For California State Parks Off-Highway Motor Vehicle Division, Sacramento, CA. 16 pp.

Grant's (2003⁴) monitoring report, plots with greater than 9% OHV track cover had lower densities of lizards. However, the primary purpose of the BLM monitoring was to estimate density to characterize an entire management area, thus sample plots were randomly selected irrespective of the lizard's habitat preferences or local precipitation levels. For this reason, confounding environmental variables likely affected the BLM's results with respect to OHV impact.

Our design featured paired plots in close proximity, assuring similarity in many environmental variables (e.g., habitat preference and precipitation levels). The USFWS, in its withdrawal of the proposed listing, requested focused and rigorous scientific research across geographic areas to aid in resolving conflicts surrounding OHV uses of *P. mcallii* habitat (USFWS 2003). Our study specifically addressed this challenge by examining differences in *P. mcallii* densities and mean *P. mcallii* mass across varying OHV impact levels, as indicated by vehicle track cover, while attempting to eliminate confounding from environmental variables.

Natural History of *Phrynosoma mcallii*

Phrynosoma mcallii has the smallest geographic range of all species in the genus *Phrynosoma* or "horned lizards" in the United States. Much of the lizard's US range is located in the Colorado Desert of southeastern California and nearby Arizona. The lizard's range extends south across the international border to northeastern Baja California and northwestern Sonora (Sherbrooke 2003, Stebbins 2003). *P. mcallii* is usually found in dry, sparsely vegetated habitat exposed to extreme temperatures. Confusing *P. mcallii* with other species of horned lizard is difficult because only the Desert Horned Lizard's, *P. platyrhinos*, range overlaps with that of *P. mcallii*. Individuals of *P. mcallii* are identified by a dark centerline along the back, a double row of fringe scales on the sides of the belly, and no spots on the belly and vent. In contrast, *P. platyrhinos* lacks a dark centerline, has only a single row of fringe scales on the sides of the belly, and has spots present on the belly and vent (Sherbrooke 2003). We encountered no adult *P. platyrhinos* during the course of this study, although one small juvenile was encountered.

Young and Young (2000⁵) reported that *P. mcallii* snout to vent length (SVL) and body mass ranged from 30 mm and 1.5 g (hatchlings) to 80 mm and 25 g (large adults). However, for this study, three individuals from Pinto Wash had SVL measurements larger than 80 mm (82, 83, and 87 mm, and 18.5, 25, and 23 g, respectively). Additionally, Wright (personal observation), who was monitoring *P. mcallii* populations at the same

⁴Wright, G. and T. Grant. 2003. Flat-tailed Horned Lizard (*Phrynosoma mcallii*) Monitoring Report. Report prepared by the Bureau of Land Management, El Centro Field Office, El Centro, CA. 57 pp.

⁵Young, K. V. and A. T. Young. 2000. Scientific Study of the Flat-tailed Horned Lizard, *Phrynosoma mcallii*. U.S. Department of Navy Contracts N68711-95-LT-C0032, N68711-95-LT-C0035. 72 pp.

time in the nearby East Mesa Management Area, recorded a SVL of 90 mm.

The diet of *P. mcallii* consists almost entirely of harvester ants, *Pogonomyrmex* sp., and *Messor pergandei*, (Norris 1949, Pianka and Parker 1975), other ant species may also be important (Turner and Medica 1982). The abundance of *P. mcallii* correlates with ant mound abundance (Turner and Medica 1982, Rorabaugh et al. 1987).

Norris (1945), Rorabaugh et al. (1987), Muth and Fisher (Desert Research Center, Univ. of California, 1992⁶), and Wright and Grant's (2003⁴) studies suggest that *P. mcallii* seems to prefer soils substrates covered in a thin layer of wind blown sand. Less frequently, the lizards inhabit other substrates, ranging from hard packed soils to dunes (Foreman 1997¹, FTHLIC 2003²) and mud hills (Beauchamp et al. 1998).

METHODS

We conducted this study from May 12 to September 5, 2003 in Imperial County, California. In an effort to eliminate confounding variables such as habitat preference and precipitation levels, we employed a random paired plot design in two separate management areas. The two study areas were oriented north and south of each other and 25 km apart. The northern study area was located in the West Mesa Management Area, established to monitor *P. mcallii* (Foreman 1997¹, FTHLIC 2003²), and the adjacent Superstition Mountain Open Area (SMOA). This northern study area included BLM lands managed as both "limited" and "open." Limited means OHV travel is restricted to officially designated routes; open means open to cross-country travel. The northern study area was located along a 17-km section of the SMOA western boundary. This study area was sparsely vegetated by the plant community of Desert Scrub (Mayer and Laudenslayer 1988). The terrain of the northern study area is largely flat, but begins to incline when approaching the Superstition Mountains to the north. Nearing the Superstition Mountains, the surface soils become sandier to the point of forming dunes at the base of these hills.

The southern study area was located within the Yuha Desert Management Area; the BLM manages the entire area as limited. More specifically, the southern study area was located in Pinto Wash along an officially-designated OHV route that follows the wash in a southwest direction, 8 km toward the international border with Mexico. The southern study area was sparsely vegetated, similar to the northern study area, but the plant communities are more diverse with species characteristic of Desert Wash (Mayer and Laudenslayer 1988), such as Honey Mesquite, *Prosopis glandulosa*, and Smoke Tree, *Dalea spinosa*. The Wash is slightly lower than the surrounding mesas; small bluffs and hills (5 to 10 m in height) outline the Wash on either side. Because the southern study area was located entirely within Pinto Wash, the surface soils are much sandier than the northern study area. For simplicity, the northern area will be referred to as West Mesa and the southern study area as Pinto Wash.

⁶Muth, A., and M. Fisher. 1992. Development of Baseline Data and Procedures for Monitoring Populations of the Flat-tailed Horned Lizard, *Phrynosoma mcallii*. Deep Canyon Desert Research Center, University of California, Riverside.

OHV impact was measured using percent cover of vehicle tracks sampled along the perimeter of each sampling plot. Management designations (open or limited) often do not determine levels of OHV track cover. For example, in Pinto Wash, high OHV track cover was found in limited areas, suggesting that management designations were not effective. Our study did not attempt to compare differences in lizard density due to management designation. Instead, our study attempted to compare differences in lizard density due to observed OHV impact.

Sample Selection

We used a paired plot random sample design to investigate possible effects of OHV impact on *P. mcallii*. Each sample consisted of two paired plots, 200 m x 200 m and separated by 300 m. One plot from each pair was designated as low OHV impact and the other as high OHV impact, based on percent OHV track cover. Before sampling was initiated, we used a geographic information system (GIS) to delineate plots (ArcGIS 8.3). In West Mesa, 42 pairs (84 plots) were delineated, of which, 30 pairs were randomly selected (Fig. 1a). In Pinto Wash, 13 pairs were delineated along the wash, starting from a random location (Fig. 1b). Because the plots were determined with a GIS beforehand, we delineated the plots in the field with a handheld global positioning system (Garmin GPS V). Percent cover of OHV tracks, sand, and perennial plants were estimated for each plot. These measures provided criteria for selection of the final three pairs in West Mesa, and the final four pairs in Pinto Wash, for a total of seven pairs.

Field technicians walked the perimeter of each plot. On every 5th pace, the technicians recorded habitat variables at a point located at the tip of the right foot. These habitat variables were soil substrate (sand, gravel, rock, or hardpan), whether a vehicle track covered the point, and whether a perennial plant covered the point. Approximately 100 points were sampled for each plot, depending on the length of the field technician's walking stride. We used only the perimeter in our estimations of percent cover because we found that the perimeter performed well. On 4 of the 14 plots, we walked the diagonals for comparison, and the difference between the perimeter and diagonals was nominal.

We used three criteria in selecting the final seven sample pairs. (1) The difference in OHV track cover between the plots was 5% or greater. Within the pair, the high impact plot contained OHV track cover of 9% or greater, and the low impact plot contained OHV track cover less than 9% (Fig. 2a). We used 9% as a threshold because Wright and Grant's (2003⁴) monitoring suggested densities of *P. mcallii* densities were reduced at this level of impact.

(2) Each plot contained 34% or greater sand cover, with a difference in sand cover of 13% or less between paired plots (Fig. 2b). We selected plots with a high percent of sand cover; as discussed above, *P. mcallii* seems to prefer a sandy soil substrate. Additionally, we attempted to select plots with similar levels of sand cover. However, a consistent but small difference in sand cover was present (Wilcoxon Signed Rank, $Z = -2.201$, $P = .028$, two-tailed). Although this difference was statistically significant, it was not practically significant for the purposes of this study, because sand cover was

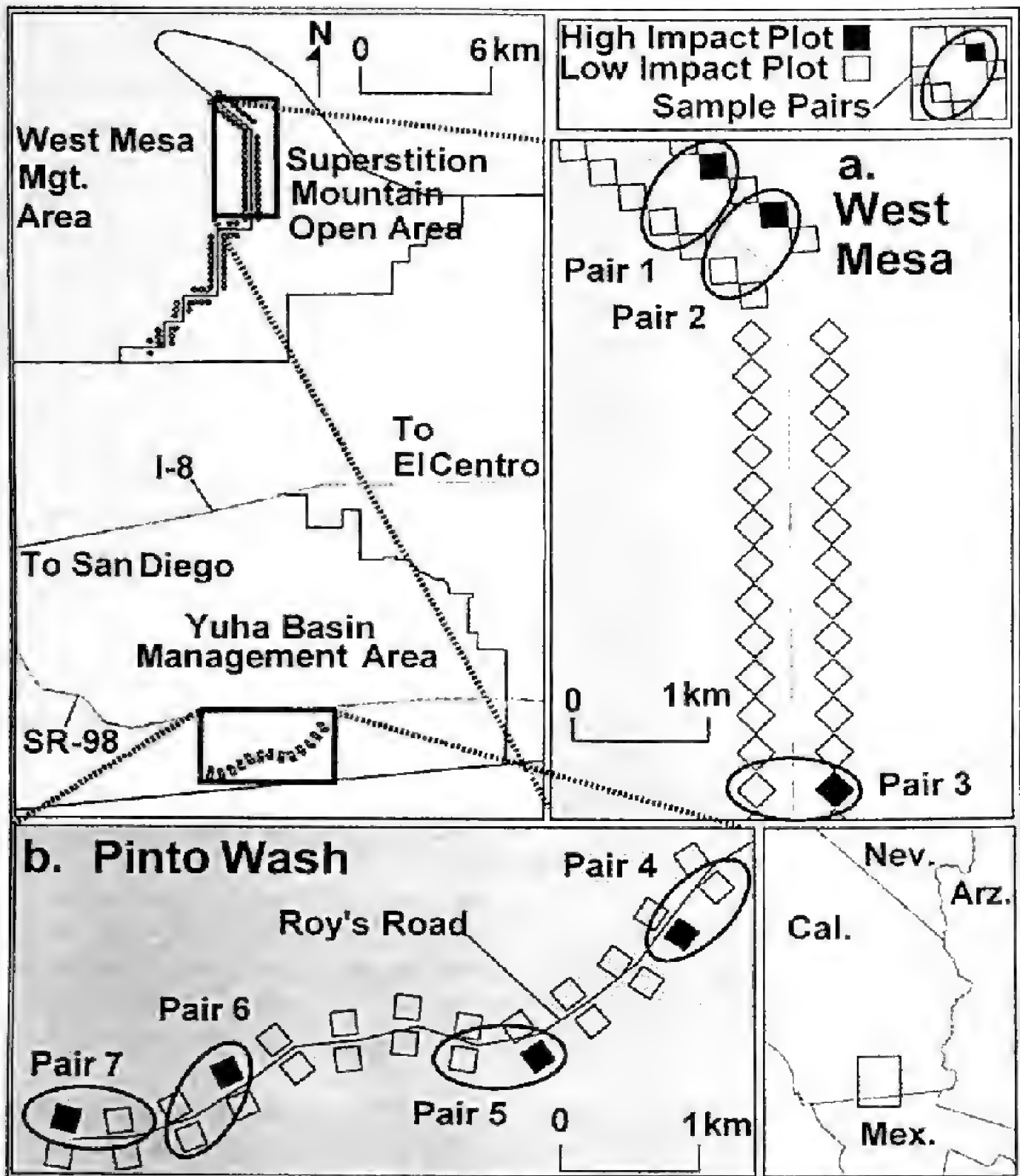


Figure 1: Map displaying the location and spatial design of the West Mesa (a) and Pinto Wash (b) study areas.

high for both plots and the difference between plots was small. In other words, both the high and low impact plots had a high percent cover of the preferred soil substrate (thin layer of wind blown sands) relative to the other paired plots that did not meet this criterion. The small, but statistically significant difference may be the result of the

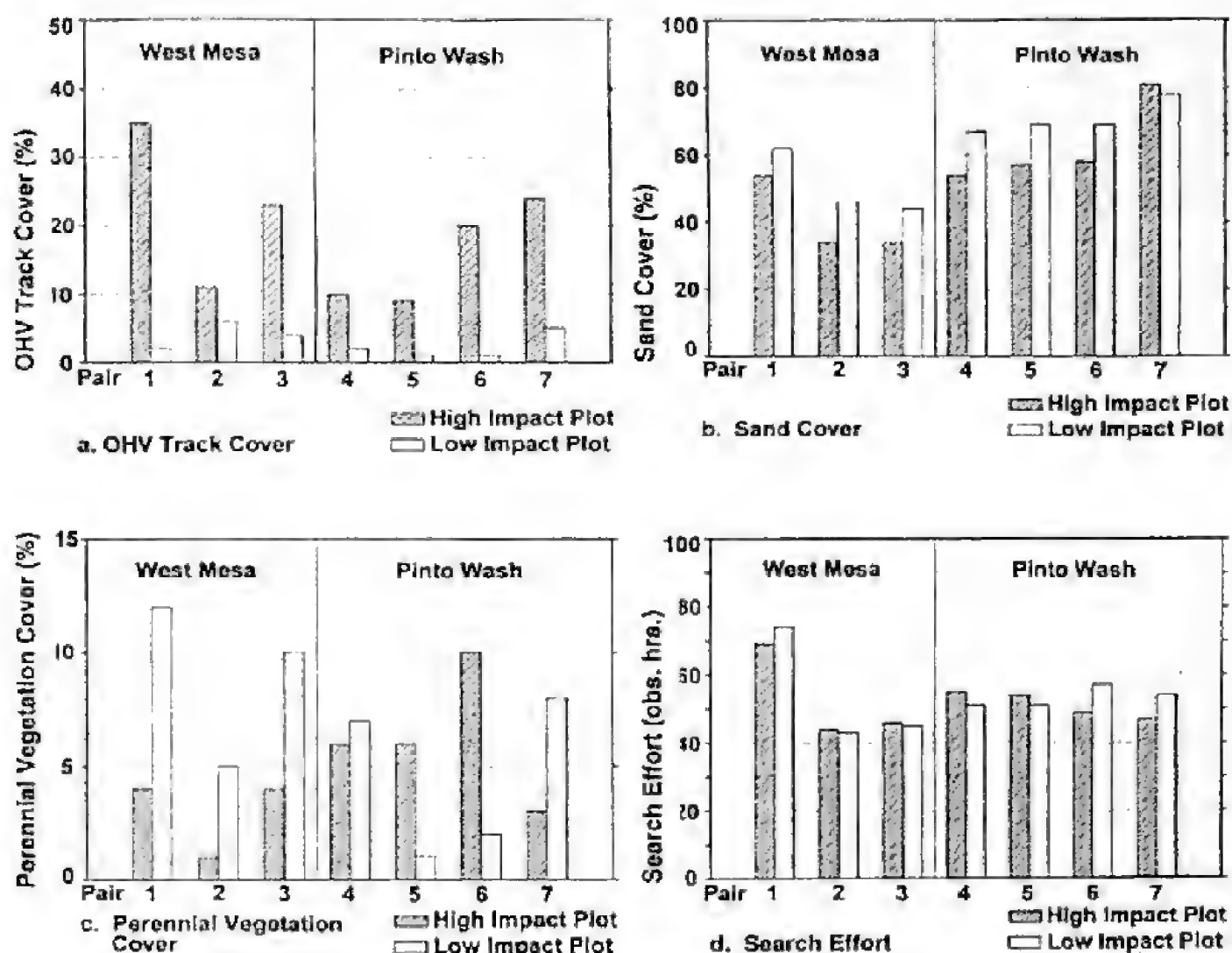


Figure 2: Percent cover of vehicle tracks (a), sand (b), perennial vegetation (c), and search effort in observer hours (d) on high and low impact plots in both West Mesa and Pinto Wash study areas.

compaction of soils due to OHV impact (Webb and Wilshire 1983, Foreman 1997¹, and Lovich and Bainbridge 1999, FTHLICC 2003²), which was related to the very question that we examined.

(3) The difference in vegetation cover was less than 8% (Fig. 2c); OHV impact may reduce vegetation cover (FTHLICC 2003²). Despite the effect of OHV impact on perennial vegetation cover, we attempted to select pairs with similar cover. The three pairs chosen in the northern study area, and the four pairs in the southern study area satisfied all three criteria.

Density Estimates

For all 7 pairs (14 plots), observed *P. mcallii* and active ant mounds were recorded with 5 consecutive search occasions per plot. Approximately 10 field days were required to complete each pair, and paired plots were sampled subsequently on the following week. We employed a mark-recapture method to produce an observed and

estimated lizard density for each plot. A search occasion consisted of three trained observers beginning at dawn and searching for 3 to 4 hours. The three observers searched the substrate while slowly walking abreast in the same direction down 20, 10-m wide lanes. Field technicians spent similar observer effort (observer hours/plot) for each paired-plot (Fig. 2d). The lanes and the perimeter of the plot were delineated with alternating colors of pin flags, in order to assure that in subsequent search occasions the entire plot received equal search effort. When encountering a lizard, the technician simply grasped the lizard barehanded and a variety of data were collected on the individual. The lizard's mass (g) was recorded using a portable scale (Pesola). SVL (mm) was recorded using a ruler. If the lizard was not marked from a previous search occasion, then it was marked on the belly, using a permanent ink marker, with the identification number of the plot and a serial number.

Other data were recorded such as sex, geographic coordinates of the encounter, scat presence or absence in the lizard's digestive track (by touch), temperature 1 cm above the surface of the substrate at the location of the encounter, surface temperature of the body of the lizard (Raytek infrared gun), and the time of the encounter. These data were not used in our analysis but may be valuable for future studies.

Active ant mounds were totaled for each plot over the five search occasions to produce an observed ant mound density (ant mounds/plot). An ant mound was counted regardless of the species and size of the mound and only if ants were entering or exiting the mound. Active ant mounds were marked with pin flags to prevent double counts on subsequent search occasions.

We produced two types of lizard density for each plot from mark-recapture techniques: observed and estimated. Observed lizard density was the number of different individuals observed from all five search occasions. Estimated lizard density was produced using mark-recapture data in the null model (Otis et al. 1978) of Program Capture, a component from the mark-recapture analysis software POPAN-5 (Arnason et al. 1998⁷). The null model is the simplest model, in which the probability of capture and the probability of recapture are held constant.

Additionally, mean lizard mass was calculated for each plot. Mean lizard mass was the mean mass of all individuals observed on each plot after five search occasions.

Data Analysis

We used the nonparametric Wilcoxon Signed Ranks test to compare the difference in ranked distributions for plots paired by OHV track cover as high vs. low impact. The Wilcoxon Signed Ranks test was applied to the following variables: (1) observed and, (2) estimated lizard density, (3) observed densities of adults (SVL \geq 60mm) and juveniles (SVL < 60mm), and (4) active ant mound density. The one-tailed Wilcoxon

⁷Arnason, A. N., C. J. Schwarz, and G. Boye. 1998. POPAN-5: A data maintenance and analysis system for mark-recapture data. Scientific Report, Dept. of Computer Science, Univ. of Manitoba, Winnipeg. [Online, URL: <http://www.cs.umanitoba.ca/~popan/>]

Signed Ranks test was chosen because of the small sample size, and the inability of the data to meet the assumptions (e.g., normal distribution of differences) of the paired-sample t-test. We analyzed the data separately by study area and with the data combined from both study areas, although qualitative differences exist between the two study areas in OHV impact and habitat. The null hypothesis was that no difference in the above variables existed between paired plots.

We analyzed a small sample ($n = 7$ pairs). However, a small sample was sufficient to detect a difference with low variability in the data. If high variability in the data was present, our ability to detect a true difference, if one existed, would have been diminished. Thus, due to our small sample size, our results were inherently conservative. For this reason, and because little previous research has been done on the affects of OHV impact specifically on *P. mcallii*, we set a more liberal alpha level ($\alpha = .1$) as our decision criterion.

RESULTS

In West Mesa plots, greater lizard densities were found in the low impact plots (Table 1). The differences between low and high impact plots in West Mesa were statistically significant, both for observed ($Z = -1.633, P = .051$) and estimated densities ($Z = -1.604, P = .055$). In contrast, a greater mean density was recorded on high impact plots in Pinto Wash for both observed and estimated densities. The differences between low and high impact plots in Pinto Wash were statistically significant for observed densities ($Z = -1.289, P = .098$), and not statistically significant for estimated densities ($Z = -1.095, P = .137$).

In West Mesa, the mean observed densities of adults ($SVL \geq 60$ mm) and juveniles ($SVL < 60$ mm) were greater on low impact plots. Observed densities of adults were not statistically significant ($Z = -.816, P = .207$) between high and low impact plots. Observed densities of juveniles were statistically significant ($Z = -1.633, P = .051$) between high and low impact plots. In contrast, high impact plots in Pinto Wash contained higher observed densities of adults and juveniles. The differences for Pinto Wash adults and juveniles were not statistically significant ($Z = -1.095, P = .137$, and $Z = 0, P = .5$, respectively).

In West Mesa, mean adult mass was greater on low impact plots, however the difference between high and low impact plots was not statistically significant ($Z = -1.069, P = .143$). Also in West Mesa, mean juvenile mass was greater on low impact plots than mean juvenile mass on high impact plots. The difference between low and high impact plots was statistically significant for mean juvenile mass ($Z = -1.604, P = .055$). In Pinto Wash, mean adult mass was higher on low impact plots than mean adult mass on high impact plots. The difference between low and high impact plots was statistically significant for mean adult mass ($Z = -1.461, P = .072$). For Pinto Wash juveniles, mean juvenile mass was greater on low impact plots than mean juvenile mass on high impact plots, however the difference between low and high impact plots was not statistically significant ($Z = -.535, P = .297$).

Table 1: The mean () and standard deviation (SD) of both low and high impact plots (N = number of pairs) for each variable is shown separated by study area (West Mesa or Pinto Wash) and combined. * indicates plot with greater mean.

		Low Impact			High Impact		
Study							
Areas		SD	N				N
West Mesa	Observed density (lizards/plot)	4.33*	1.53	3	1.67	1.53	3
	Estimated density (lizards/plot)	6.67*	2.52	3	1.67	1.53	3
	Adult density (lizards/plot)	2.33*	0.58	3	1.67	1.53	3
	Juvenile density (lizards/plot)	2.00*	1.73	3	0.00	0.00	3
	Active ant mound density (mounds/plot)	19.33*	11.72	3	12.33	3.22	3
	Mean adult mass (g)	15.42*	0.52	3	10.14	8.83	3
	Mean juvenile mass (g)	5.65*	5.94	3	0.00	0.00	3
Pinto Wash	Observed density (lizards/plot)	7.25	4.35	4	11.75*	5.74	4
	Estimated density (lizards/plot)	9.00	5.42	4	15.00*	7.87	4
	Adult density (lizards/plot)	5.00	4.24	4	8.50*	6.35	4
	Juvenile density (lizards/plot)	2.25	2.63	4	3.25*	4.27	4
	Active ant mound density (mounds/plot)	32.50*	26.76	4	24.25	19.16	4
	Mean adult mass (g)	17.66*	5.07	4	12.15	8.04	4
	Mean juvenile mass (g)	2.35*	8.14	4	0.90	1.03	4
Combined	Observed density (lizards/plot)	6.00	3.56	7	7.43*	6.80	7
	Estimated density (lizards/plot)	8.00	4.28	7	9.29*	9.09	7
	Adult density (lizards/plot)	3.86	3.34	7	5.57*	5.86	7
	Juvenile density (lizards/plot)	2.14*	2.12	7	1.86	3.14	7
	Active ant mound density (mounds/plot)	26.86*	21.29	7	19.14	15.08	7
	Mean adult mass (g)	16.70*	3.79	7	11.29	7.77	7
	Mean juvenile mass (g)	3.76*	4.51	7	0.51	0.87	7

Active ant mound densities (mounds/plot) were calculated for each plot. In West Mesa, observed densities of active ant mounds were higher on low impact plots than on high impact plots. The same pattern was observed in Pinto Wash, where ant mound densities were higher on low impact plots. The difference between low and high impact plots was not statistically significant in West Mesa ($Z = -1.069, P = .143$), but statistically significant in Pinto Wash ($Z = -1.289, P = .099$).

Additionally, the combined data from both West Mesa and Pinto Wash was analyzed. Mean adult mass, mean juvenile mass, and active ant mound densities were

greater on low impact plots for the combined study areas. The differences between low and high impact plots for mean adult mass ($Z = -1.69$, $P = .045$), mean juvenile mass ($Z = -1.782$, $P = .037$), and active ant mound densities ($Z = -1.609$, $P = .054$) were statistically significant. However, the differences between low and high impact plots for observed density ($Z = -.594$, $P = .276$), estimated density ($Z = -.423$, $P = .336$), and observed density of adults ($Z = -.512$, $P = .304$, one-tailed) and juveniles ($Z = -.954$, $P = .170$) was not statistically significant.

DISCUSSION

Overall, our results from West Mesa supported the hypothesis that densities, body mass, and food resources are lower in high impact plots, but in Pinto Wash this hypothesis was not supported with respect to densities. We offer two explanations of these differences in West Mesa and Pinto Wash: (1) qualitative differences in OHV impacts existed between the study areas due to management conditions (open vs. limited), and (2) methods used for measuring OHV impacts were sensitive enough to detect a difference in West Mesa, but were not sensitive enough in Pinto Wash.

Qualitative Differences

The differing pattern observed between West Mesa and Pinto Wash, could be explained by qualitative differences in the study areas, particularly in OHV management. In West Mesa, one plot of each pair was chosen from an open management area and one plot from a limited area. In contrast, both plots of each pair in Pinto Wash were selected from a limited OHV management area. The effect of OHV impacts on desert environments vary, depending on the type of OHV impact (FTHLICC 2003²). West Mesa largely receives impacts from recreational OHV activity, while Pinto Wash largely receives impacts from the non-recreational OHV impact of the US Border Patrol and illegal immigrant traffic (USFWS 2003, Wright personal communication).

Recreational OHV impact in West Mesa was often concentrated around camping and staging areas that are re-used season after season, while non-recreational OHV impact in Pinto Wash was often sporadic in time and space. The measure of OHV track cover on plots in West Mesa may not be analogous to Pinto Wash because the measure does not include cumulative impacts over time. In sum, for a given measure of OHV track cover, OHV impact in West Mesa may actually be more severe than Pinto Wash OHV activity.

In addition to differing types of OHV activity, West Mesa and Pinto Wash differed in habitat qualities. *P. mcallii* density was substantially higher in Pinto Wash than West Mesa, perhaps because Pinto Wash substrate, perennial plant diversity, and active ant mound density were more suitable for the lizard (Turner and Medica 1982, FTHLICC 2003²). The population in Pinto Wash may be more resilient to more sporadic and less intense OHV impacts because of higher *P. mcallii* densities and more favorable habitat characteristics.

Lizard Mass and Ant Mound Densities

The results are consistent with the hypothesis that OHV impact reduces *P. mcallii* mass. Lizards had greater individual body mass on low impact plots in both West Mesa and Pinto Wash. The diet of *P. mcallii* consists almost entirely of ants (Pianka and Parker 1975, Turner and Medica 1982). Differences in active ant mound densities were statistically significant when the combined data from both study areas were analyzed. High impact plots contained lower active ant mound densities for both West Mesa and Pinto Wash. As a result, high impact areas may be poorer habitat for satisfying nutritional needs of *P. mcallii*. Additionally, lizards may have been distressed, expending more energy in avoidance behavior as a result of disturbance caused by OHV activity, or spending more time motionless as a response to perceived danger of OHV impact (Nicolai and Lovich 2000). Either way, the lizards may not be feeding while OHV impact is occurring nearby. Additionally, lizards may cover larger distances in search of food sources resulting in larger home ranges (Wone and Beauchamp 2003).

RECOMMENDATIONS

Our results suggest that recreational OHV impact may reduce lizard densities and that recreational and non-recreational OHV impact may reduce individual lizard body mass. We recommend increased protection of *P. mcallii* habitat from OHV impact in order to protect this unique animal from further potential population and fitness declines while future studies determine the extent of OHV effects. We suggest that OHV impact be measured over a time interval, rather than at a single moment in time, perhaps across a recreational season and preferably multiple seasons. Additionally, the effects of recreational and non-recreational activity should be differentiated in future studies because of qualitative differences in frequency, intensity, and management. We also recommend geographically expanding the scope of the study areas. This could improve external validity of future studies, making results applicable to different locations. This study cannot make causal inferences from its comparative results. A study that experimentally manipulates OHV impact would indicate causality. However, although experimental measurements may be more sensitive, manipulated OHV impact may not be analogous to authentic OHV impact because of temporal and spatial differences. In this differential study, we attempted to measure authentic OHV activity. We propose that additional experimental studies are necessary to validate the suggested impacts from this study of authentic OHV activity.

Based on the results of the study we recommend the following management practices. (1) *P. mcallii* management areas should be closed to OHV activity, not simply managed as limited, until further research has been conducted on the effects of OHV activity. The results of West Mesa, Pinto Wash, and both study areas combined imply that even moderate amounts of OHV impact may negatively affect *P. mcallii* body mass and fitness. (2) If OHV impact must be allowed in the management areas, then the activity should be confined to only a few routes, and these routes should be well marked and enforced. (3) OHV impact associated with camping and staging areas should not

occur in otherwise good lizard habitat. In the Algodones Dunes of southeastern California, Wright (1999⁸) reported to the BLM that Colorado Desert Fringe-toed Lizard, *Uma notata*, abundance declined on transects when nearing camping and staging areas. In our study, camping occurred in good habitat in West Mesa, which may explain the differences observed there. In sum, we recommend increasing protection of *Phrynosoma mcallii* habitat from off-highway vehicle activity.

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⁸Wright, G. 2003. Colorado Desert Fringe-toed Lizard (*Uma notata*) Abundance in Relation to Vehicle Impacts at the Algodones Dunes, 1987 to 1999. Report prepared by the Bureau of Land Management, El Centro Field Office, El Centro, CA. 11 pp.

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INFLUENCE OF SURFACE-WATER WITHDRAWAL ON JUVENILE STEELHEAD AND THEIR HABITAT IN A SOUTH-CENTRAL CALIFORNIA NURSERY STREAM

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INTRODUCTION

Tributaries and the nursery habitats they possess have a critical role in the ecology of steelhead, *Oncorhynchus mykiss*. Adults migrate into tributaries, where characteristics and conditions are well suited for production of young (Erman and Hawthorne 1976, Montgomery et al. 1999). Parr that emigrate from upstream natal areas during spring to rear in mainstem habitats often return to tributaries in the fall (Hartman and Brown 1987, Bramblett et al. 2002), presumably as an adaptation to environmental conditions for increasing the chance of survival. The presence of diverse age classes of steelhead in tributaries (Roper et al. 1994) indicates the capability to support life history characteristics such as multi-year freshwater residence (Chapman 1958, Narver 1969). Given the functional value of tributaries, protection of such habitats should receive high priority.

The exploitation of surface-water and ground-water resources in coastal basins of south-central California is common and has created much concern for fishery managers. Habitat changes due to streamflow alterations can negatively affect the physicochemical and biological properties of streams (Poff et al. 1997) and are believed to have contributed to the population decline of anadromous salmonids (Hedgecock et al. 1994, Moyle 1994). The ecological concern over such alterations has been sufficient to prompt the American Fisheries Society to render policy regarding this issue (policy statement 9) and considerable efforts to restore stream function and fishery resources (Smith et al. 2000). Given the interest in conserving anadromous salmonids (Nehlsen et al. 1991), understanding how water diversions affect fishery resources is a necessary step for developing reliable mitigation strategies.

The influence of streamflow alterations on anadromous fishes, particularly as related to large water-storage projects (Blahm 1976, Raymond 1988, Mundie 1991), has been documented. However, at smaller scales the influence of surface-water withdrawal and habitat changes on juvenile steelhead, especially in the absence of instream storage facilities, is poorly understood. Without such information, predictions regarding the possible influences of such activities on steelhead and their habitat are tenuous; this uncertainty translates into a risk, particularly for threatened and endangered species. In this note, we report the findings of a study performed in summer 2003 that examined instream habitat characteristics and density of juvenile steelhead (the South Central California Coast Evolutionarily Significant Unit) upstream and

downstream of a surface-water pump in a small south-central California tributary stream.

STUDY AREA

See Canyon Creek (35°11'19"N, 120°43'10"W) is a tributary to San Luis Obispo Creek, which itself is a tributary to the Pacific Ocean (Fig. 1). The surrounding area is rural, with a few scattered private residences. Oak, *Quercus* spp., woodland dominates the surrounding hills, whereas riparian areas possess willow, *Salix* spp., alder, *Alnus* spp., and California sycamore, *Platanus racemosa*. Pool, run, and glide are common, and we observed numerous short riffles. A few prickly sculpin, *Cottus asper*, were observed in the study area. Extensive accumulations of large (≥ 10 -cm diameter) and small woody debris are common in the creek channel, the bed of which is mostly cobble and gravel. The depth and width of pools sampled for this study averaged 0.2 and 2.6 m, respectively. Adult steelhead spawn in the study area, as indicated by observation of actively spawning fish and numerous redds. Additional information regarding the San Luis Obispo Creek watershed, including See Canyon Creek, can be found in Spina et al. (2005).

The study reach extends 400 m upstream (reference reach) and 400 m downstream (affected reach) of a surface-water pump, which is located 0.8 km upstream of the intersection of San Luis Bay Drive and See Canyon Road (Fig. 1). The length and boundaries of the study reach were selected based on instream habitat characteristics and the expected downstream extent of effects owing to pump operations, based on preliminary surveys. A diesel generator stationed on the creek bank operates a pump (capacity of 68 l L/min) to irrigate grazing land for cattle. The pump intake is screened to exclude fish and is located in the middle of the channel at the bottom of a pool. Based on interviews with the landowner (who operates under a riparian water right) pumping began on or about 26 May and ended on 3 August 2003, in response to a cease-and-desist order issued by the California Department of Fish and Game. During this period, the landowner generally pumped water from the creek 6 days a week, 14 to 16 continuous hours each day, beginning at 1600 hours and ending at 0600 hours.

METHODS

The study reach was inspected on 7 July to note characteristics of the instream habitat (e.g., wetted channel width, whether surface water was present and flowing) and to estimate stream discharge in affected and reference reaches. Stream discharge was measured before (at two locations in the affected reach, during 1550-1615 hours) and after (in two locations within the affected and reference reaches, 1941-1955 hours, and 1847-1909 hours) pumping began using the method for streamflow measurement described in Platts et al. (1983¹) and an electronic current meter, which was calibrated

¹Platts, W.S., W.F. Megahan, and G.W. Minshall. 1983. Methods for evaluating stream, riparian, and biotic conditions. U. S. Forest Service General Technical Report INT-138.

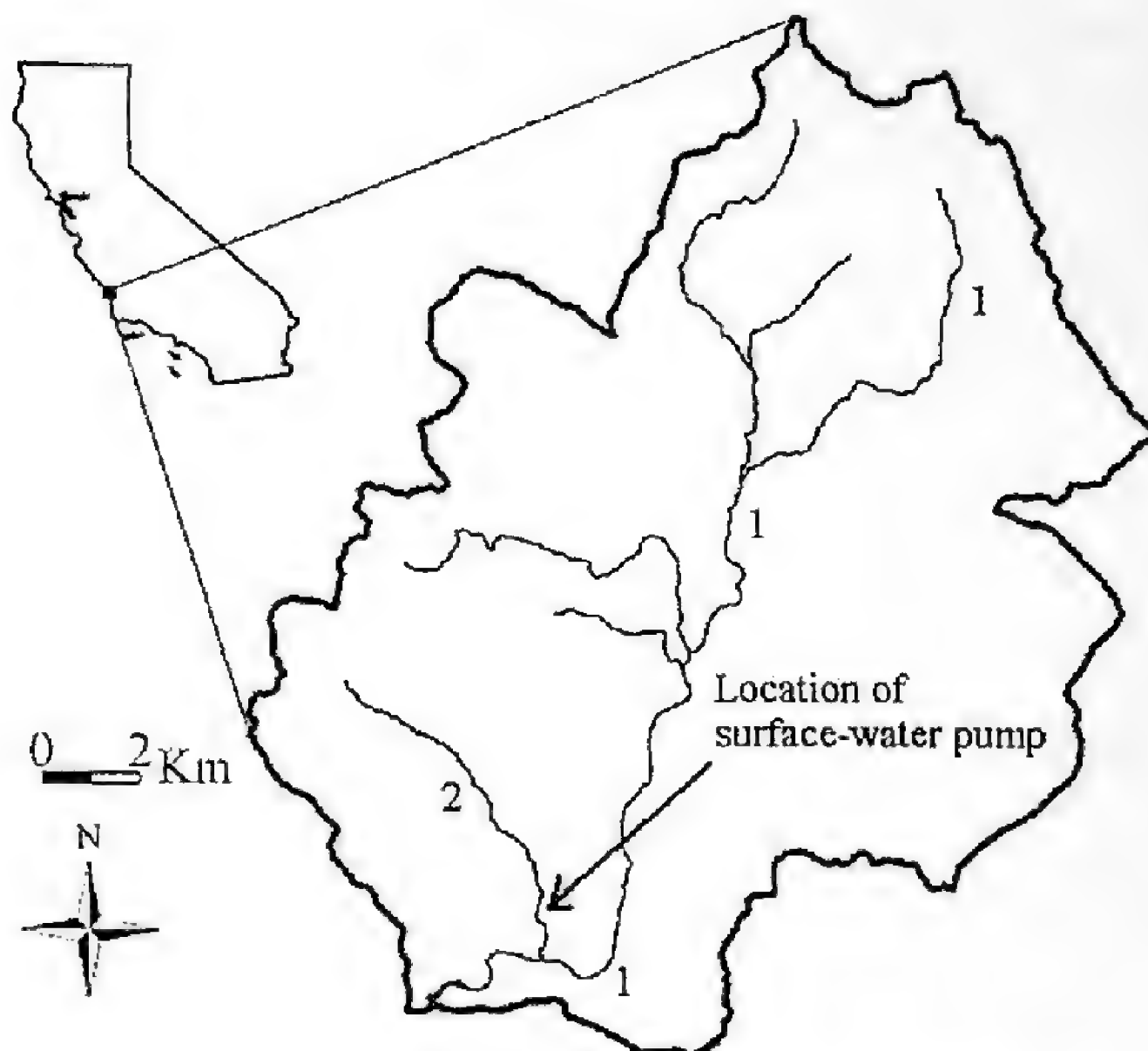


Figure 1. San Luis Obispo Creek watershed showing San Luis Obispo Creek (#1 on map), See Canyon Creek (#2), other principal streams, and the location of the surface-water pump. The study area extended 400 m upstream and 400 m downstream of the pump.

prior to use according to manufacturer guidelines. Because measurements during preliminary surveys indicated that pumping did not reduce discharge in the reference reach, we use the measurements of stream discharge taken in the reference reach after pumping began to characterize discharge for this reach. The measurements of stream discharge were averaged for each reach and time period. A similar visual inspection of the instream characteristics was performed during a pumping event on 28 and 30 July, but streamflow measurements were not repeated.

To estimate pool-specific mean density of juvenile steelhead upstream and downstream of the pump, 9 pools were sampled within both the reference and affected reaches (total of 18 pools). The first nine pools located upstream and downstream of the subject pump were selected for sampling, except two particular pools located upstream of the pump were excluded from sampling. Capturing fish in these specific

pools using electrofishing was likely to be extremely challenging and inefficient due to the extensive accumulations of woody debris. The mean depth and surface area (\pm 1SD) of the pools sampled were 0.2 (0.05) m and 34.9 (13.6) m², respectively in the reference reach, and 0.2 (0.5) m and 27.0 (9.1) m², respectively in the affected reach. These measurements were taken when the pump was not operating (during 4 to 7 August).

During 4 to 7 August, we used the removal-depletion method (Riley and Fausch 1992) and one backpack electroshocker to collect steelhead from the 18 pools. To ensure that no fish left or entered the sample unit during the survey, we enclosed each sample unit with block nets, and used cobbles and small boulders to secure the net lead line to the channel bed and eliminate voids between the net and channel bed. Other than installing block nets at unit boundaries, we did not disturb the sample unit prior to electrofishing. A minimum of three passes with the electroshocker was performed in an upstream direction and the entire area of each sample unit (including steelhead sheltering areas such as cobbles and woody debris) was systematically shocked during the first pass and each successive pass to maintain constant effort among the passes. After electrofishing a unit, the depth (m) and width (m) of the sample unit were measured at randomly selected locations using a random-numbers table and the *x-y* coordinate method (Whalen and Parrish 1999).

Steelhead were measured (mm, fork length) and scales were removed from 43 individuals prior to release for the purposes of characterizing the age-at-length relationship for the collected individuals and then generating age-specific density estimates. The specific lengths of steelhead in which scales were removed for analysis were chosen based on our knowledge of the age-at-length relationship for juvenile steelhead in the region (Spina 2003), and we removed scales from a slightly broader range of fish lengths (i.e., surrounding an expected age break) than was suggested by our prior knowledge. Length-frequency distributions were prepared to identify two age categories (age-0, and age-1 and older) (Fig. 2), and the age breaks in the distribution were validated with scale readings using standard methods (Jerald 1983).

MicroFish 3.0 (Van Deventer and Platts 1989²) was used to compute maximum-likelihood estimates of fish abundance for the two age categories in each sample unit. When steelhead were captured only during the first and not successive passes, MicroFish sets the population estimate for the subject unit to the total catch because maximum-likelihood estimation is not possible in this situation. Pool-specific density estimates (number of fish/m²) were calculated for each age category using the estimate of fish abundance divided by the area of each sampled unit. The fact that we captured few fish in some pools should not decrease confidence in our findings for at least a few reasons. First, we believe fish did not leave sample units during the electrofishing surveys owing to the block nets. Second, we maintained constant effort among the passes by, in part, ensuring that the entire sample unit (including locations that did not

²Van Deventer, J.S., and W.S. Platts. 1989. Microcomputer software system for generating population statistics from electrofishing data—user's guide for MicroFish 3.0. U.S. Forest Service, General Technical Report INT-254, Ogden, Utah, USA.

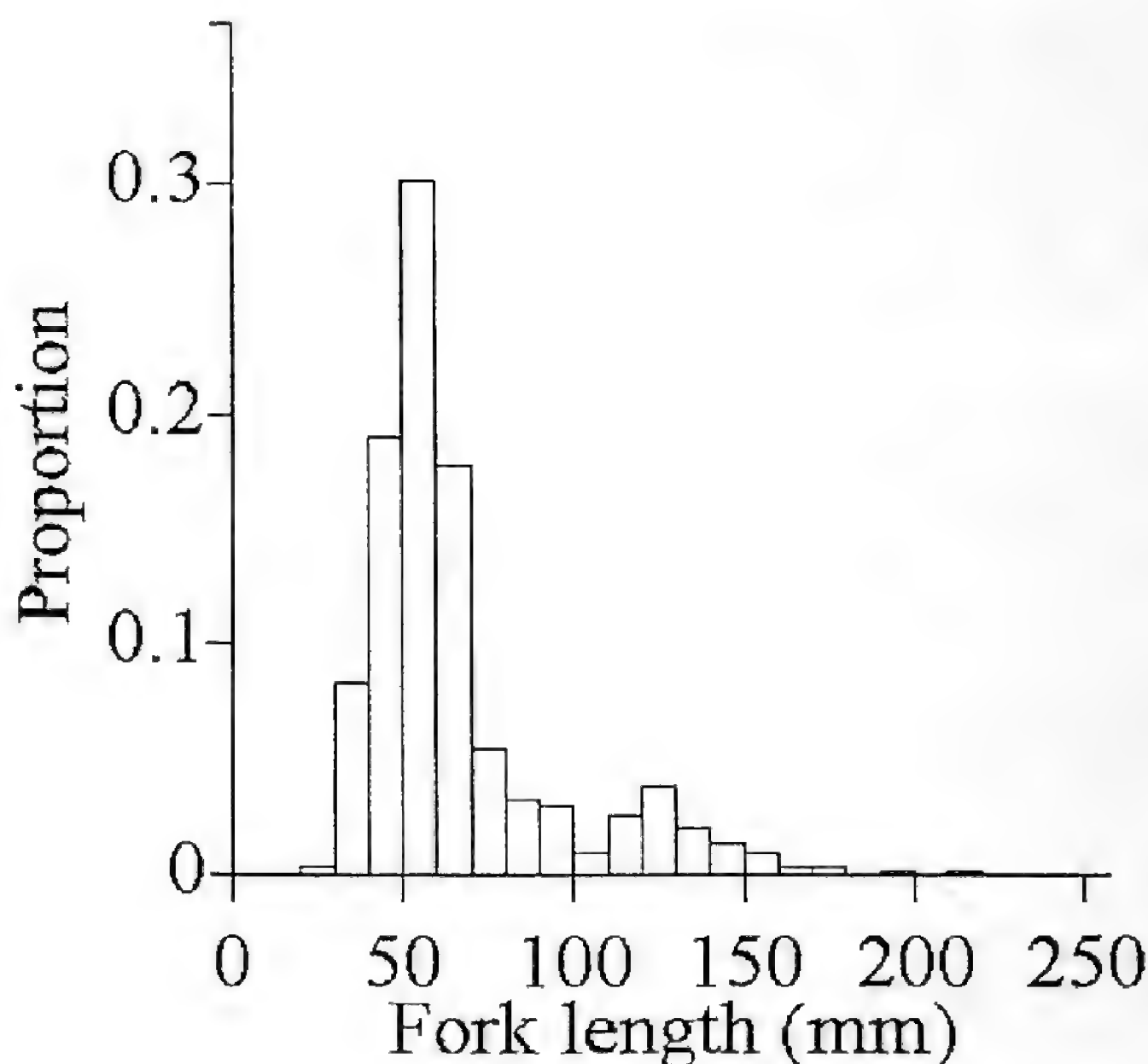


Figure 2. Length-frequency distribution of steelhead collected from pools in the study area of See Canyon Creek during 4 to 7 August 2003 ($n = 672$). The break between age-0 and age-1 and older steelhead was assigned at 100 mm and was validated with results of scale readings.

produce fish on a previous pass) was thoroughly shocked. Third, we constantly monitored electrofishing voltage and amperage, and when necessary rotated new batteries into the surveys, to maintain electrofishing efficiency. Fourth, conditions were ideal for electrofishing; discharge was relatively low and water clarity was unlimited even in the deepest pools. Overall, the fact that we collected few juvenile steelhead in some pools is a reflection of the low abundance of fish in the subject pools.

RESULTS

After pumping began on 7 July, flowing water within the reach extending 100 m downstream of the pump was eliminated, resulting in dewatered runs, riffles, and glides

and pronounced reductions in pool depth (on average 7.6 cm) within this specific area. Although surface water persisted farther downstream, reductions in wetted width ranged from 50.8 cm to 91.4 cm (Fig. 3). Before pumping, estimated discharge in the affected reach was 2.8 L/s, then decreased to 1.4 L/s after pumping began and was much less than the reference discharge (8.5 L/s). On 28 and 30 July, we again observed that flowing water was lacking within the reach extending 100 m downstream of the pump, and instream habitat had been dewatered (i.e., little to no surface water). Two dead steelhead were found in the dewatered reach on 30 July.



Figure 3. Example of the reduction in wetted channel width observed during July 2003 inspections. Arrows show wetted width before (long arrow, 119 cm) and during (short arrow, 45 cm) withdrawal of surface water.

During the electrofishing surveys, we collected a total of 672 juvenile steelhead, 413 individuals from the reference reach and 259 from the affected reach. Mean density (\pm 1SD) of age-0 steelhead was 0.8 (0.9) fish/m² in the affected reach and 1.2 (0.5) fish/m² in the reference reach. Mean density of age-1 and older steelhead was 0.1 (0.2) fish/m² in the affected reach and 0.1 (0.1) fish/m² in the reference reach. Densities of both age groups were lowest in affected pools nearest the pump, and in the case of age-1 and older steelhead, no individual was collected or observed in the four affected pools nearest the pump (Fig. 4).

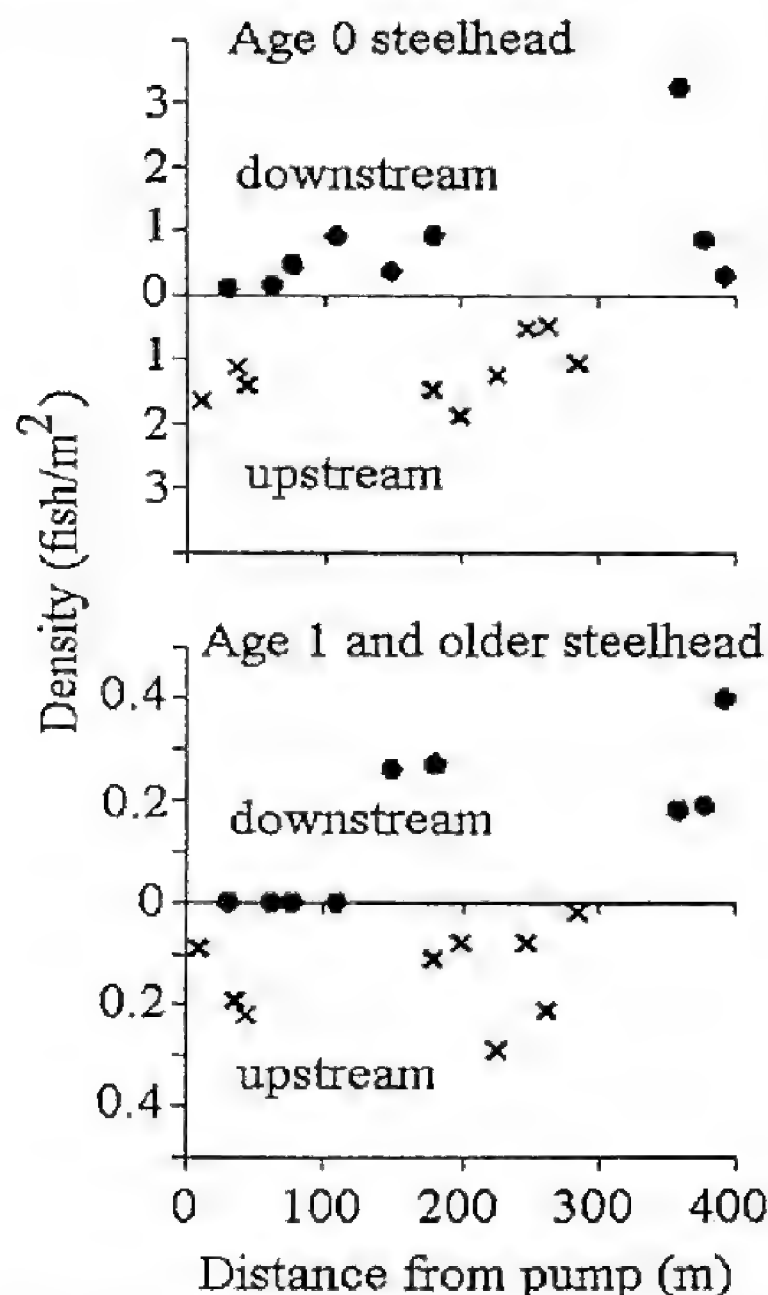


Figure 4. Relationship between density of age-0 and age-1 and older steelhead in pools upstream (reference reach) and downstream (affected reach) of the pump, and distance from the pump (located at distance zero).

DISCUSSION

Our findings corroborate those of other investigators reporting decreased abundance of aquatic organisms downstream of water-withdrawal locations where habitat quality and quantity have been reduced (Vinyard 1996, McIntosh et al. 2003). In our study, we neither collected nor observed age-1 and older steelhead in the four affected pools nearest the pump, which were within the localized area where the most notable reductions in living space, and the dead steelhead, were observed. The reduction in living space probably exposes steelhead (and possibly other aquatic organisms that we did not consider) to abiotic and biotic conditions that are unfavorable for survival and growth (Cushman 1985). The loss of depth alone may reduce the chance of survival,

especially for larger juveniles, based on the value of deep water for bigger fish (Power et al. 1989, Harvey and Stewart 1991).

We do not know whether a difference in steelhead density between the two reaches existed prior to the pumping, but we did not detect a difference in mean density between reaches, only a decreased density of steelhead (e.g., age-1 and older individuals) in affected pools nearest the pump was documented. Although the physical characteristics of the four pools offer an alternative hypothesis for the absence of age-1 and older steelhead within the localized area, our site-specific observations and general knowledge of habitat use in juvenile steelhead (e.g., Spina 2003) lead us to believe that physical characteristics of the four pools (including the amount and type of cover) were suitable for older juveniles and were similar to those of other sampled pools. Two pools possessed substantial overhanging and instream branches that provided shade and shelter, and area and mean depth of the four pools were similar to other sampled pools. The physical characteristics of the four pools seem unlikely to fully explain the absence of age-1 and older juvenile steelhead.

We conclude that pump operation created unfavorable instream conditions that lead to the absence of age-1 and older steelhead in affected pools nearest the pump. Although landowners possessing a riparian water right have certain entitlements to stream water, our findings indicate the fishery resource and their habitat were not adequately considered, perhaps not at all, in the execution of that right. The results of our study underscore the need to ensure anthropogenic activities are sufficiently evaluated and balanced against the habitat requirements of anadromous salmonids, even when the scale and potential consequence of an activity are perceived as small or negligible.

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**NEW RECORDS OF THE FLAG CABRILLA,
EPINEPHELUS LABRIFORMIS (SERRANIDAE:
EPINEPHELINAE), FROM THE PACIFIC COAST OF BAJA
CALIFORNIA, MEXICO, AND SAN DIEGO, CALIFORNIA,
USA, WITH NOTES ON THE DISTRIBUTION OF OTHER
GROUPERS IN CALIFORNIA**

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The flag cabrilla, *Epinephelus labriformis* (Jenyns 1840), is a carnivorous marine fish in the family Serranidae. This species is taken in artisanal and commercial fisheries throughout its range (Ramírez and Rodríguez 1990; Craig et al. 1999), and usually found over hard substrate where it feeds on small fishes and crustaceans (Hobson 1968; MTC, DJP, RNL, personal observation). Typical of other species in the family, the flag cabrilla may reach ages in excess of 25 years despite a maximum size of only 500 mm standard length (Craig et al. 1999). The pelagic larval stage of the flag cabrilla may remain in the plankton for up to 50 days (B. Victor Ocean Science Foundation, and MTC unpublished data). Recent evidence suggests that it is a protogynous hermaphrodite, switching gender from female to male at approximately 400 mm standard length (B. Erisman, Scripps Institution of Oceanography, and MTC, unpublished data).

Widespread throughout the Tropical Eastern Pacific (TEP), the flag cabrilla has been recorded from the Gulf of California to Peru, including the offshore islands of Revillagigedo, Coco, Malpelo, and Galápagos, at depths of up to 30 m (McCosker and Rosenblatt 1975; Heemstra and Randall 1993). Although previously reported from Clipperton Atoll (Heemstra and Randall 1993; Allen and Robertson 1994), the flag cabrilla is not present at this locality. Individuals previously identified as *E. labriformis*

at Clipperton Atoll have recently been described as *E. clippertonensis* (Allen and Robertson 1999). The flag cabrilla is abundant in the central portion of its range, gradually reaching lower densities in the northern Gulf of California (MTC, RNL, DJP personal observation). Herein, three new records of *E. labriformis* are reported from Cedros Island, San Benito Islands (Baja California, Mexico), and San Diego, California. These previously unreported records are discussed, indicating that this species should be added to the ichthyofauna of the outer Baja coast and southern California.

On 2 November 1998, DJP observed a specimen of *E. labriformis* on the leeward side of Cape San Augustine, Cedros Island, BC, Mexico, while scuba diving. The specimen was noted as exhibiting typical behavior for the species, perched over rocky substrate in 12 m of water. The temperature of the water was 16 °C. On a subsequent dive trip, RNL and DJP observed a solitary *E. labriformis* at the San Benito Islands on 8 August 2001. The individual was approximately 300 mm total length, and was observed in 10 m of water. The water temperature was 18.3 °C at the surface. An additional diver observation of this species was recorded from San Benito Islands in 1989 (RNL personal observation).

Recently (October 2004), a specimen of *E. labriformis* was collected at Rocas Alijos, Baja California, Mexico, by Dr. Russ Vetter. The specimen was 325 mm SL, and was collected in approximately 15 m of water by hook and line. The specimen is deposited in the Scripps Institution of Oceanography Marine Vertebrates Collection (SIO 04-178) and confirms earlier records of this species at the Rocas Alijos (SIO 70-371).

While cataloged museum records of *E. labriformis* are numerous throughout the southern portion of its range, relatively few records have been reported along the outer (Pacific) coast of the Baja California Peninsula (Table 1). One of these records represents a substantial, unreported range expansion to the north (SIO 71-170; Figure 1). This 88-mm SL (104 mm TL) specimen was brought to the Vaughan Aquarium at Scripps alive by a local fisherman, who indicated he had caught the specimen at the Mission Bay Jetty (32°47.0' N, 117° 15.0' W) in September 1971. The identification of the specimen was originally performed by R. H. Rosenblatt and recently confirmed by MTC and RNL. Counts for the specimen are: D XI, 16; A III, 8; Pect. L 17, R 18; GR 7 upper, 15 lower, 22 total. The collection locality was presumed to be accurate, although some doubt remained as is apparent from the notes of Dr. Rosenblatt, Curator Emeritus, written in the catalogs of the SIO Marine Vertebrates Collection:

"Specimen brought to SIO Aquarium by man who claimed to have caught it at the above locality [Mission Bay Jetty]. His name was not taken and aquarium staff did not immediately realize importance of the record, so it was not and can not ever be confirmed by further documentation. Specimen was kept alive for two months and then preserved."

Given that the specimen was brought alive to the Vaughan Aquarium, and in light of the difficulties that would be associated with transporting a specimen in such a condition across the United States-Mexico boundary (i.e., long distance to transport a live fish), there is little or no reason to doubt the validity of the collection locality.

Many records of *E. labriformis* are available from the area of Bahía Magdalena

Table 1. Museum records of *Epinephelus labriformis* from the Pacific Coast of Baja California, Mexico, and southern California, USA. Institutional abbreviations follow Leviton, et al. (1985).

Locality	Museum Number
Cabo and Bahía San Lucas	CAS-SU 17586 SIO 59-210 SIO 59-215 SIO 61-225 SIO 61-227 SIO 61-232 SIO 61-233 USNM 4247*
Rocas Alijos	SIO 70-371 SIO 04-178
Punta Tosco	SIO 64-57 SIO 64-58
Bahía Magdalena	CAS-SU 23 SIO 60-309 SIO 60-386 SIO 62-96 SIO 62-105 SIO 62-131 SIO 62-711 SIO 62-724 SIO 64-52 SIO 64-53 SIO 65-54 SIO 65-55
Bahía Santa Maria	SIO 60-312 SIO 62-734 SIO 64-42
Mission Bay, California	SIO 71-170

*Holotype of *Epinephelus sellicauda* Gill, 1862. This nominal species is a junior synonym of *E. labriformis* (Jenyns 1840).

(Table 1). One specimen (CAS-SU 23) was collected in a beam trawl aboard the U.S. Fish Commission steamer "ALBATROSS" in 1889. The specimen was taken in 27 fm of water off Bahía Magdalena, BCS, Mexico. The presence of adult *E. labriformis* on the outer coast of the Baja California Peninsula during the late 1800s suggests that this species may regularly inhabit the temperate/sub-tropical waters of the area and has done so in limited numbers, thus going relatively unnoticed in scientific collections. Between 1853 and 1860, numerous Panamic fishes correlated with a warming trend were reported in this region (Hubbs 1948). The noted faunal shift in the late 1970s from the Oregonian to San Diegan species assemblage (Stephens et al. 1986) initiated by the

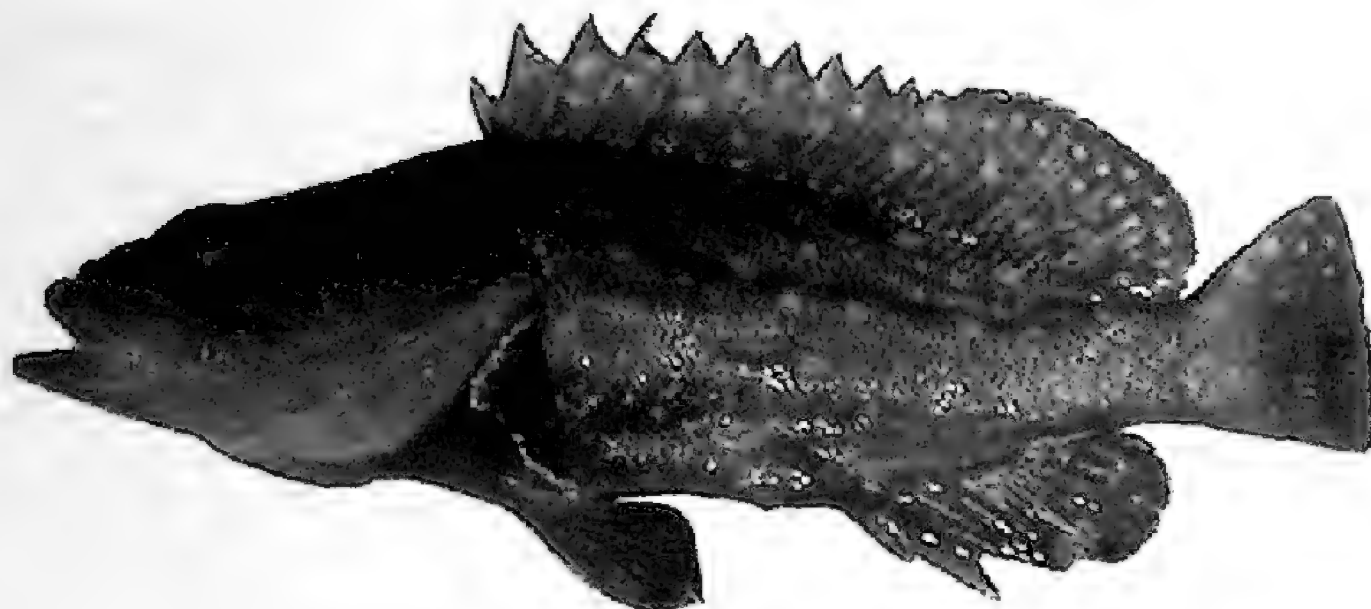


Figure 1. Photograph of SIO 71-170, *Epinephelus labriformis*, from Mission Bay Jetty, San Diego, California. The specimen is 88 mm standard length.

strong El Niño Southern Oscillation (ENSO) event of 1978-79 marked a change to persistently warm oceanographic conditions in the southern California/northern Baja California area. This change allowed for species with broad Panamic distributions to move northward (e.g., Brooks 1987; Lea and Fukuhara 1991; Pondella 1997; Pondella et al. 1998; Lea and Rosenblatt 2000). The limited data available for marine fish distributions on the outer coast of the Baja Peninsula prior to this shift does not allow for a robust conclusion to be drawn regarding the historical distribution of *E. labriformis*. The 88-mm SL specimen taken at the Mission Bay Jetty (SIO 71-170) may have reached its extreme northward location by larval transport during the 1969-70 ENSO event. The small size indicates that it was most likely between 1 and 2 years of age upon capture (Craig et al. 1999) which is consistent with this hypothesis.

There is no indication that this species has ever been present in the aquarium trade, thus ruling out the possibility of accidental or intentional release into the San Diego area by aquarists. While there was (and still remains) an active long-range fishing fleet both in San Diego and Mission Bay, we do not feel that this species could have been introduced into the area by "hitchhiking" on such a vessel as an adult or larva; most live wells in such vessels are usually flushed well prior to their return to port. While a possibility does exist that larger ships carrying ballast water into the San Diego Bay may have inadvertently transported the specimen as a larva or sub-adult, we feel that the likelihood of this is small, and that the individual most likely recruited to the area.

Southern California is noted for the continued presence of some other epinepheline serranid species whose affinities lie with the tropical and sub-tropical waters of the eastern Pacific. The broomtail grouper, *Mycteroperca xenarcha*, is reported to occur as far north as San Francisco Bay (Miller and Lea 1976) and the species is regularly observed in King Harbor, Redondo Beach, on the federal breakwater of Long Beach and Los Angeles Harbors, and in the La Jolla kelp (Stephens et al. 1994, Pondella 1997;

Table 2. Catalogued museum records for groupers (Epinephelinae, Serranidae) in California. Institutional abbreviations follow Leviton et al. (1985).

Species	Catalog Number	County	Locality	Date
<i>Dermatolepis dermatolepis</i>	LACM 44895	Los Angeles	Scattergood Generating Station	1988
<i>Epinephelus acanthistius</i>	LACM 44983	Los Angeles	Santa Monica Bay, Redondo Canyon	1989
<i>Epinephelus analogus</i> ¹	SIO 60-176	San Diego	La Jolla	1960
<i>Epinephelus niphobles</i>	LACM 36292	Los Angeles	Redondo Beach	1975
<i>Epinephelus niphobles</i>	LACM 38381	San Luis Obispo	Point Buchon - Avila	1974
<i>Epinephelus niphobles</i>	LACM 38417	San Luis Obispo	Point Piedras Blancas	1975
<i>Epinephelus niphobles</i>	LACM 44148	Los Angeles	Santa Monica Bay	1986
<i>Epinephelus niphobles</i>	LACM 44243	Los Angeles	Redondo Beach	1975
<i>Epinephelus niphobles</i>	LACM 45563	Los Angeles	Santa Monica Bay	1992
<i>Epinephelus niphobles</i>	SIO 65-8	San Diego	NW of North Coronado Island	1965
<i>Epinephelus niphobles</i>	SIO 74-74	San Luis Obispo	Estero Point	1974
<i>Mycteroperca jordani</i>	SIO 45-151	San Diego	La Jolla	1945
<i>Mycteroperca jordani</i>	SIO 45-152	San Diego	La Jolla	1945
<i>Mycteroperca jordani</i>	SIO 45-225	San Diego	La Jolla	1945
<i>Mycteroperca jordani</i>	SIO 51-319	San Diego	La Jolla	1951
<i>Mycteroperca jordani</i>	SIO 55-54	San Diego	La Jolla	1955
<i>Mycteroperca xenarcha</i>	CAS 58469	Monterey	Monterey Bay	1986
<i>Mycteroperca xenarcha</i>	SIO 45-151	San Diego	La Jolla	1945
<i>Mycteroperca xenarcha</i>	SIO 45-152	San Diego	La Jolla	1945
<i>Mycteroperca xenarcha</i>	SIO 47-107	San Diego	Solana Beach	1947
<i>Mycteroperca xenarcha</i>	SIO 48-300	San Diego	Pacific Beach - Birdrock	1948
<i>Mycteroperca xenarcha</i>	SIO 50-147	San Diego	La Jolla	1950
<i>Mycteroperca xenarcha</i>	SIO 56-12	San Diego	Birdrock	1956
<i>Mycteroperca xenarcha</i>	SIO 66-297	San Diego	Solana Beach	1966

¹ An additional California record of the spotted cabrilla, *E. analogus*, was reported on by Lea and Rosenblatt, 2000.

J. Froeschke Occidental College, DJP, MTC. personal observation). Sporadic reports of six other grouper species also exist, and verified museum specimens confirm their presence in central and southern California (Table 2).

While these species are fairly uncommon in California waters today, there existed a moderate sized fishery for grouper, *Mycteroperca jordani*, and *M. xenarcha*, and spotted cabrilla, *Epinephelus analogus*, in the early to mid 20th century off the southern coast of Baja California (Fitch 1949). In 1930, the landings of cabrilla, *Epinephelus analogus*, totaled 500,000 lb and in 1941 the reported landings of grouper, *Mycteroperca* spp., totaled 280,000 lb (Fitch 1949). While these landings of grouper and cabrilla were reported in California, the main fishing grounds were further south off of the southern Baja Peninsula (Fitch 1949).

Most museum records indicate that these species were adults at the time of capture, suggesting that they were either residents that had settled from larvae at that locality or had migrated as adults to these more northerly sites. In the case of *E. niphobles*, which has been collected as far north as San Luis Obispo County, and *M. xenarcha*, which has been reported from San Francisco Bay (Miller and Lea 1976), it is hard to accept the latter mechanism of transport. The strong tendency towards reef habitat in groupers (Heemstra and Randall, 1993) necessitates structure; hence the long stretches of sandy shore characteristic of southern California's coastline (such as the region from San Diego to the Palos Verdes Peninsula and the Santa Monica Bay region) would inhibit their movement northward along the coast.

While *E. labriformis* apparently has not continued to recruit to more northern localities such as San Diego, its noted presence at various sites along the outer coast of the Baja California Peninsula warrants the addition of this species to the region's ichthyofauna. The continued presence of other grouper species in California waters remains tenuous, particularly in the face of the widespread exploitation of the group. Nevertheless, the presence and distribution of these large predatory species remains as a point of interest for further study.

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CAPTIVE BREEDING OF SPOTTED SAND BASS, *PARALABRAX MACULATOFASCIATUS*, IN SOUTHERN CALIFORNIA

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Spotted sand bass, *Paralabrax maculatofasciatus*, ranges from Mazatlán, México to Monterey, California including the Gulf of California, in depths out to 60 m (Miller and Lea 1972, Thomson et al. 2000). In southern California, spotted sand bass commonly inhabit shallow warm waters in embayments and harbors above bottom structure, predominantly eelgrass beds (Limbaugh 1955², Allen 1985, Allen et al. 1995, Hovey and Allen 2000). Unlike other members of the genus common to southern California (kelp bass, *Paralabrax clathratus*, and barred sand bass, *P. nebulifer*), spotted sand bass adults are generally restricted to coastal embayments, with little or no migration along the open coast as adults (L.G. Allen, personal observation). It has been hypothesized that southern California populations are maintained only through the dispersal of planktonic larvae along the open coast, which can lead to years of depressed recruitment when oceanic conditions are not suitable to spotted sand bass larval development, i.e. sea surface temperatures in excess of 20°C (Allen et al. 1995). Throughout their range, spotted sand bass exhibit high growth rates and early sexual maturity (< 1 year of age; Allen et al. 1995). However, they are short-lived, with most individuals dying well before their 10th year in the Southern California Bight (Allen et al. 1995).

A small, but increasing, recreational fishery for spotted sand bass exists in southern California, with landings ranging from 12,000 to over 500,000 individuals annually from 1976-1999 (Hovey and Allen 2001). The increased recreational fishing pressure and accompanying variability in spotted sand bass landings raise new issues for fishery managers. Can the spotted sand bass populations be sustained under the increasing fishing pressure within southern California? Extensive studies on this species (Allen et al. 1995, Tranah and Allen 1999, Hovey and Allen 2000, and Miller and Allen, in press) indicate that it is doubtful. First, even though spotted sand bass are fast growing and early maturing fish, they do not live long, with most dying before their 10th year. This means that fewer, older individuals exist in the populations to add continuity to the fishery. Secondly, juvenile spotted sand bass recruit to the bays from the plankton only

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²Limbaugh, C. 1955. Fish life in the kelp beds and the effects of kelp harvesting. Univ. Calif. Inst. Mar. Res., IMR Ref:55-56.

sporadically from year to year. Because recruitment is variable, large gaps in the age structure are common. Some southern California bays may go 3 to 5 years without a successful recruitment (Allen et al. 1995), which inevitably leads to population declines. Thirdly, spotted sand bass are non-migratory fish and, therefore, become isolated to the bays within which they recruit. Finally, tagging studies also indicate that most southern California populations of spotted sand bass are relatively small overall (L.G. Allen unpub. data). For example, a very high tag return rate (10%) indicated that as few as 4,000 to 10,000 spotted sand bass occurred in Newport Bay in the early 1990s. A population of that size, in any bay, cannot sustain increasing fishing pressure for long. Ironically, some of the same characteristics, namely fast-growth, early maturation, isolation, and small population size, that render spotted sand bass susceptible to overexploitation also make them excellent candidates for hatchery enhancement.

Within the state of California, several recreationally important gamefish have been the subject of considerable efforts in aquaculture, namely, white seabass, *Atractoscion nobilis* (Vojkovich and Crooke 2001) and California halibut, *Paralichthys californicus* (J. Rounds, Director, California Halibut Hatchery Program, Redondo Beach, California personal communication,). Matus-Nivon et al. (1990) suggested that spotted sand bass is suitable to culture techniques. Presently, no *Paralabrax* spp. has been cultured in southern California, but spotted sand bass have been successfully cultured in La Paz, Baja California Sur, Mexico, (Aviles-Quevedo et al. 1995, Martinez-Diaz et al. 2001). It is not known if southern California water conditions can support a viable aquaculture operation for spotted sand bass. This work tests the feasibility of a captive-breeding program in southern California for spotted sand bass.

Spotted sand bass adults were housed in two broodstock tanks at the SEA Laboratory in Redondo Beach, California (33°50.50'N, 118°23.70'W). Each tank was a round 1100-l indoor, insulated fiberglass tank supplied by a flow-through seawater system. Fish were fed frozen northern anchovies, *Engraulis mordax*, daily. Tanks were cleaned daily a minimum of 1 hour after feeding to remove fecal matter and uneaten food. The water in each tank was heated by submersible 300-w heaters to maintain a mean water temperature > 20°C from April to December. Both tanks were illuminated with 120-w plant lights (specialized bulbs designed to imitate natural solar wavelengths) timed to regulate a photocycle of 14L:10D starting on 9 May 2002. From April 2002 to June 2002, 36 fish collected from San Diego Bay, California (32°37.30N, 117°14.80W) by hook and line, were housed in one tank. On 17 June 2002, 10 fish were moved from tank one (T1) into tank two (T2), while 26 individuals remained in T1. Both groups were maintained through June 2003. High natural mortality, of unknown causes, was encountered in spring 2003, necessitating the addition of six fish to T1 and eight fish to T2 and in July 2003. The new fish were collected from Mission Bay, California (32°45.54'N, 117°15.38'W) by hook and line with ambient water temperature of 24°C.

The first confirmed spawning event occurred on 18 May 2002. Egg collectors (500-micron mesh bag) were affixed over the end of the outfall pipe of each tank to capture eggs. Each collector was emptied into a 20-l bucket filled with aerated seawater and allowed to settle for 20 minutes to separate viable (live, fertilized, relatively clear) eggs from dead or unfertilized eggs (cloudy or black coloration) and detritus. Viable eggs

remained buoyant at the water's surface while all remaining material would sink out. Egg production was estimated volumetrically each day for each broodstock group's reproductive output.

Spawning during the first trial (May to December 2002) continued on a near daily basis until December 2002 (Figure 1), with a total mean daily egg production of 38 ml eggs/day. Water temperatures generally remained above 19°C for the duration of the spawning period in 2002 (Figure 1). The two groups had highly variable levels of daily egg production, ranging from 0 ml to 200 ml in group one and 0 ml to 75 ml in group two (Figures 2 and 3). Combined daily mean egg production peaked in July (> 70 ml eggs/day total), which corresponded to the absolute peak in mean monthly water temperature over the course of the study (Table 1). August was the next most productive month for each group, with a relative decline in egg production through the end of summer 2002 (Figures 4 and 5). Egg production in excess of 5 ml was measured through October 2002 (Figures 4 and 5). Mean monthly egg production was significantly correlated with mean monthly water temperature for the T1 group ($r = 0.89$, $P = 0.018$) and the T2 group ($r = 0.93$, $P = 0.008$), from July–December (Figures 4 and 5). In the second trial (May 2003 to October 2003), egg production dropped to a mean of 1.6 ml eggs/day while daily water temperatures remained in a similar range (19.0–22.8°C) as was measured in the first trial. Little or no spawning output was measured in either group during the second trial from the individuals collected from Mission Bay, California. Spawning output by the individuals from Mission Bay, California may have been depressed by the decrease in ambient water temperature they were exposed to, 24°C at time of collection to a mean of approximately 20°C in the broodstock tanks.

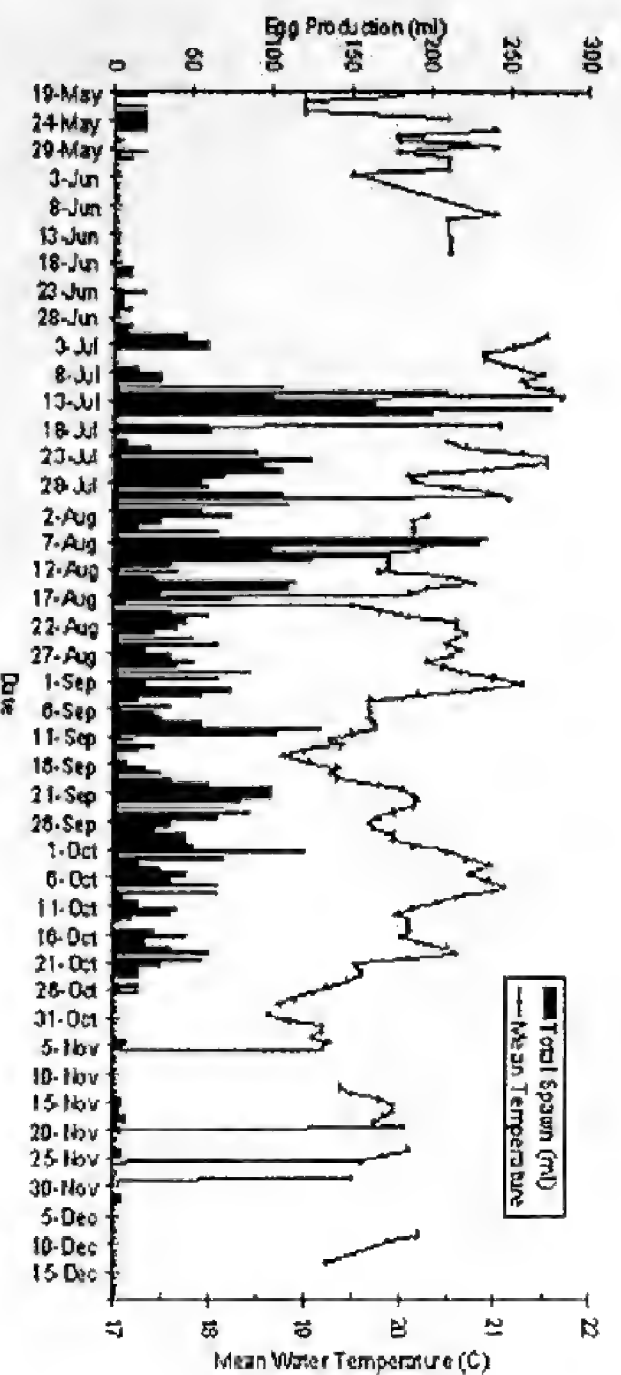


Figure 1. Mean daily water temperature and total egg production by all 36 captive adult spotted sand bass, 2002.

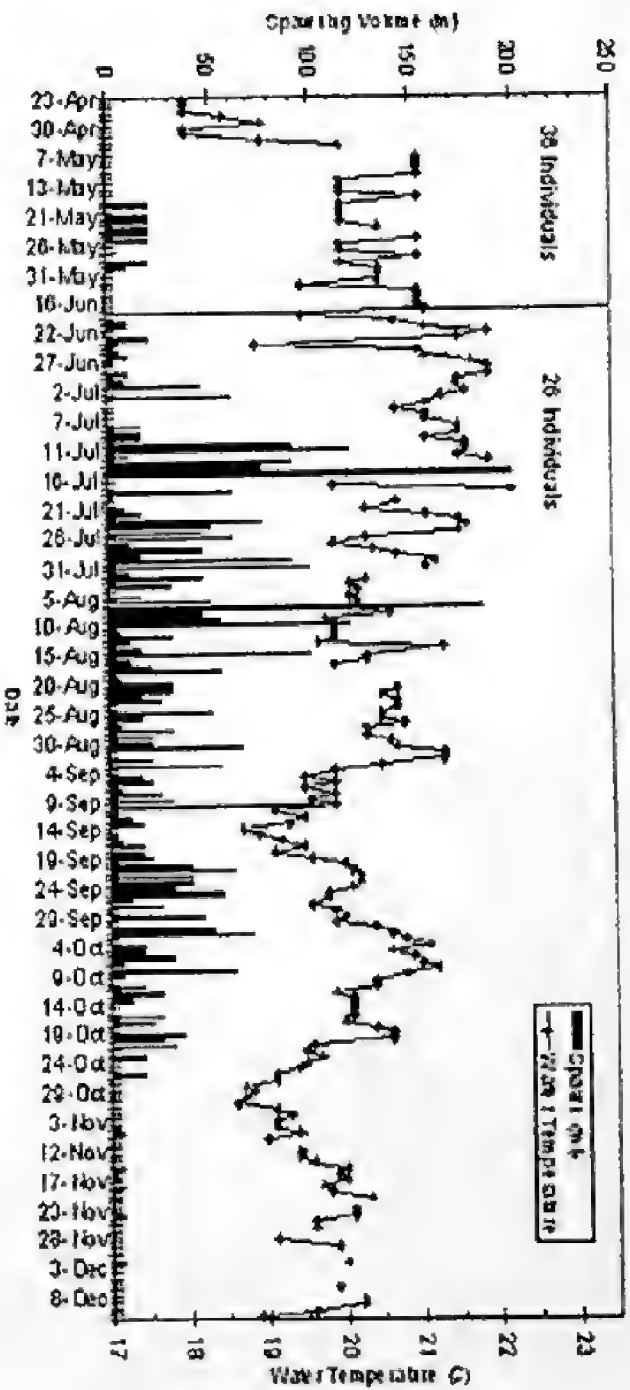


Figure 2. Daily water temperature and egg production for the group in tank one, 2002.

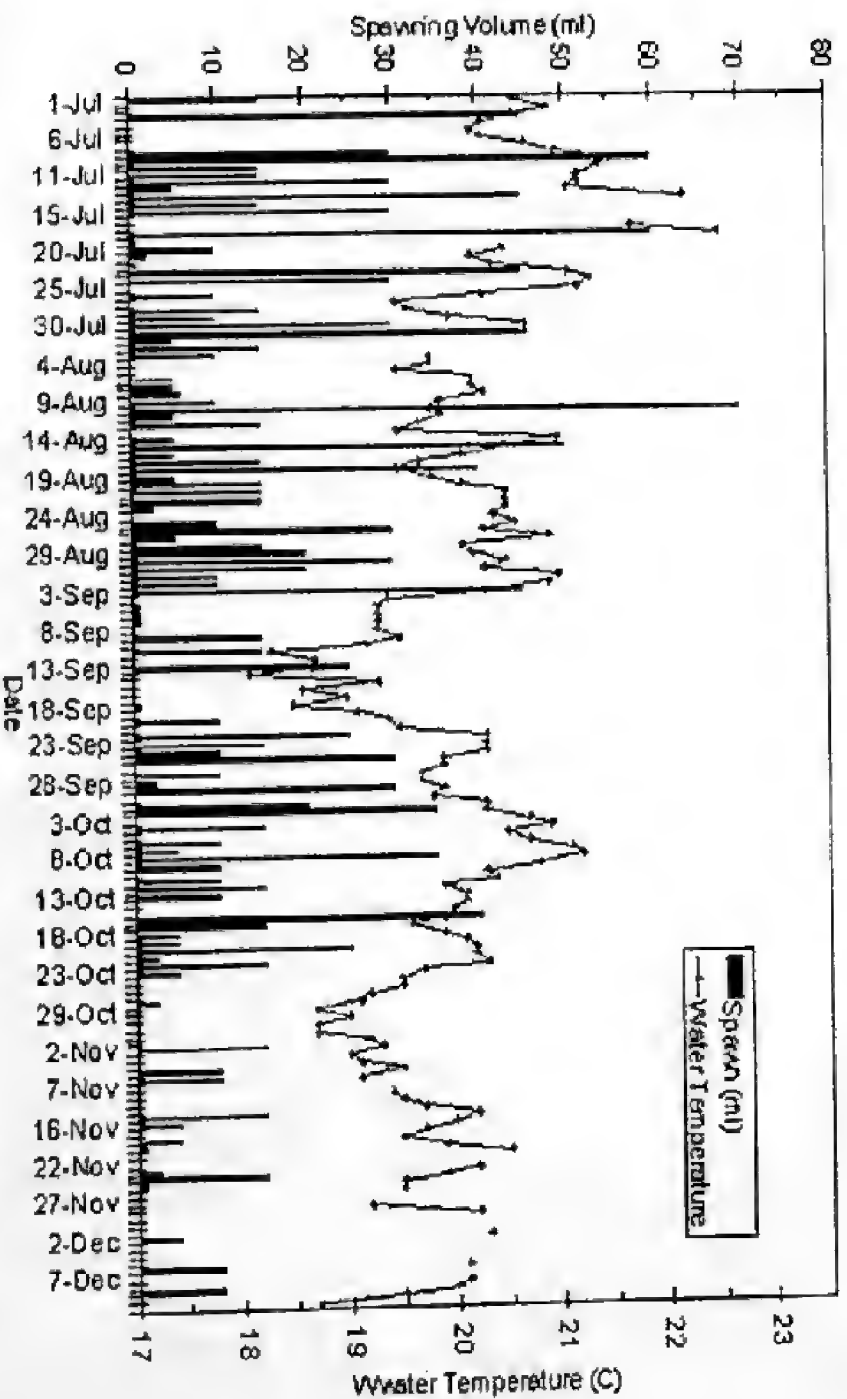


Figure 3. Daily water temperature and egg production for the group in tank two, 2002.

Table 1. Monthly mean egg production in milliliters and water temperature in °C for the broodstock spotted sand bass.

Month	Tank 1 mean egg production (ml)	Tank 1 mean water temperature °C	Tank 2 mean egg production (ml)	Tank 2 mean water temperature °C
April	0.00	18.3	NA	NA
May	11.07	20.2	NA	NA
June	5.00	20.9	NA	NA
July	50.33	21.1	21.69	20.1
August	37.00	20.3	14.78	20.8
September	24.14	19.8	8.84	19.5
October	20.22	20.0	12.17	20.0
November	2.03	19.6	5.30	19.6
December	1.00	19.8	6.25	19.8

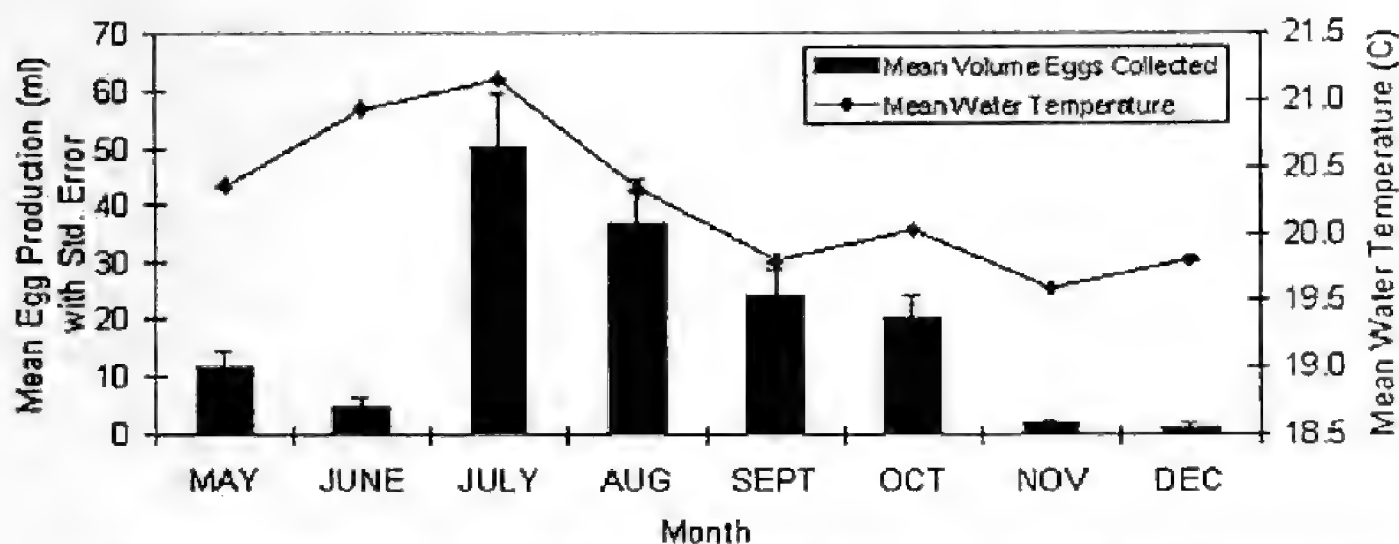


Figure 4. Mean monthly egg production and water temperature for the tank one group, 2002.

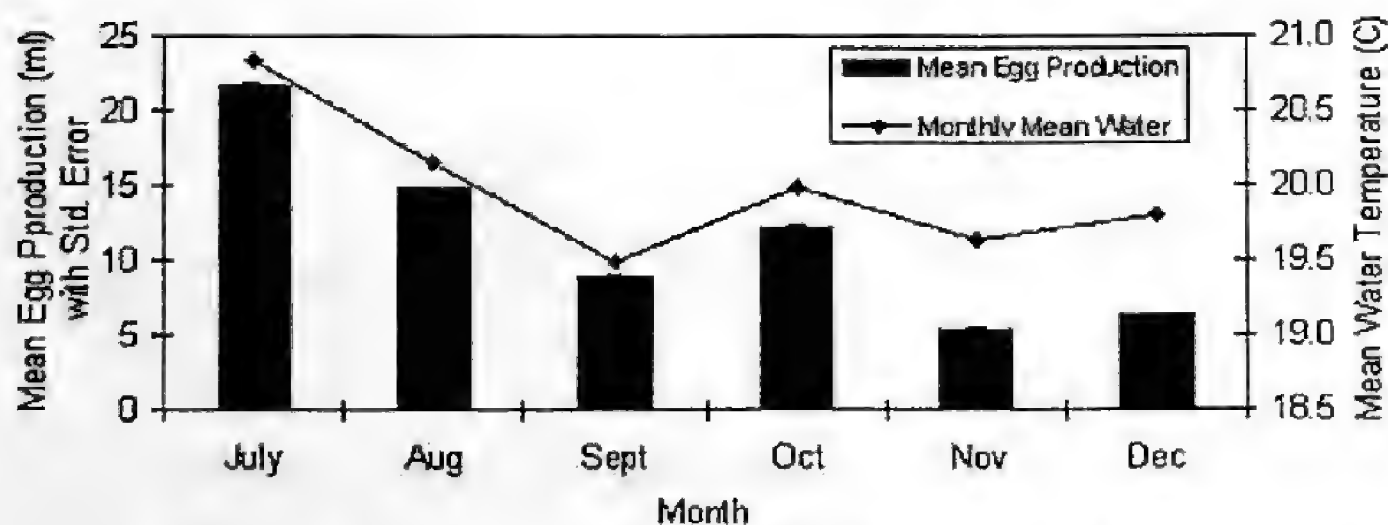


Figure 5. Mean monthly egg production and water temperature for the tank one group, 2002.

Eggs hatched within 24 h of collection and were placed either in indoor 500-l circular tanks or outdoor 700-l circular tanks. Indoor tanks were filled with 20- μ m filtered seawater and aerated, while outdoor tanks were filled with unfiltered seawater and aerated. Water temperature was not regulated in the larval grow-out tanks. In the first trial, grow-out tanks were stagnant with daily water changes during cleaning. During the second trial, tanks had flow-through systems with a flow rate of 4-l/hr and cleaned once weekly (D. Weaver, Owner, Scientific Hatcheries, Huntington Beach, California, personal communication).

During the first trial, larvae were fed artificial diets, while the second trial used live rotifers, *Brachionus* sp., raised on microalgae, *Nannochloropsis* sp., through 20 days after hatch (DAH). After 20 DAH, live *Artemia* sp. nauplii were added to the diet. Both rotifers and *Artemia* sp. were introduced to the tank either by automatic feeders set to feed four times daily, or by continuous drip. After 40 DAH, larvae and juveniles were fed fish flesh, such as northern anchovy, *Engraulis mordax*. During the first trial, one individual survived to settlement (35 DAH) before perishing. In the second trial, seven individual fish survived through transformation into juveniles.

In summary, the present hatchery efforts produced eight post-settlement juveniles, with three surviving past 6 months of age. These juveniles were cultured from eggs obtained from the captive broodstock via voluntary spawning. Although two types of food (artificial and live) were used in successive years, only live prey diets successfully supported the growth and development of the larvae through transformation into juveniles, following settlement. The larvae raised on artificial diets during the first trial showed little growth.

The seven juveniles (Figure 6) produced in 2003 were the first spotted sand bass cultured in southern California from a captive broodstock to survive past 40 DAH. This result, while very modest, does indicate that a hatchery program for spotted sand bass warrants further work as a potential resource to enhance the natural stocks, particularly in years of low recruitment among wild stocks. Increased refinement of culture techniques for spotted sand bass could allow for direct supplementation of the natural stocks. Due to the limited migration and increasing fishing pressure, stock enhancement activities for spotted sand bass may provide benefits to the fishery similar to those observed in freshwater systems, i.e. trout stocking programs.

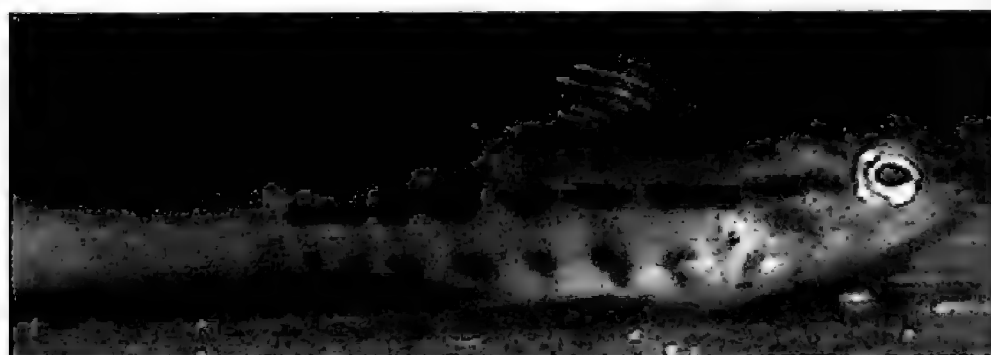


Figure 6. Juvenile spotted sand bass at approximately 80 days after hatch, 2003.

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DISTRIBUTION OF TUI CHUB IN THE COW HEAD BASIN, NEVADA AND CALIFORNIA

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Cowhead Lake tui chub, *Gila bicolor vaccaceps*, is endemic to the Cow Head sub-Basin, Warner Basin, California (Bills and Bond 1980). Like many native fishes of Western North America, it has been negatively impacted by habitat alteration (Minckley and Douglas 1991). Loss of historic habitat and its current limited habitat led to a proposed rule for listing on March 30, 1998 (U.S. Fish and Wildlife Service 1998); however, federal and state agencies and landholders developed a conservation agreement and strategy to reduce threats to Cowhead Lake tui chub and promote its long-term survival, thus obviating listing. A principle objective of the conservation agreement was to develop baseline information needed to determine the appropriate and most feasible conservation actions. Our study, conducting a survey to better define the range of tui chub in the Cow Head Lake Slough region of the Warner Basin, is an essential component to future conservation of the Cowhead Lake tui chub.

Cow Head Lake is situated in the southern Warner Basin, California, and is flanked by the Warner Mountains (maximum elevation within Cow Head watershed 2341 m mean sea level [m.s.l.] to the west and volcanic plateau and low mountains (maximum elevation 1925 m m.s.l.) to the east. Cow Head Lake bed covers approximately 1829 hectares at 1593 m m.s.l.; the watershed drains 10,384 hectares. Runoff originates mostly from snowmelt, which historically was conveyed by its three primary tributaries, Eightmile, Ninemile, and Barrel creeks (Figure 1). The Lake discharged into Cow Head Slough, then Twelvemile Creek, which joined Twentymile Creek and fed a chain of lakes in Warner Valley. After the area was settled, water was manipulated to irrigate agricultural land in and out of the immediate drainage basin, and to dry the Lake bed so it could be used for agricultural purposes. A portion of the water from Eightmile Creek watershed is diverted via an excavated channel to Surprise Valley Basin to the south, where it is stored in Annie Reservoir and subsequently used for irrigation. Likewise, the discharge of Barrel Creek is rerouted from Cow Head Lake to Cow Head Slough. Even with these diversions, runoff still enters Cow Head Lake in late winter and early spring and must be pumped before the lake bottom can be available for agricultural use. Cow Head Ditch (a man-made channel) approximately 1 km long and 10 m wide at the northwest end of Cow Head Lake and a large diesel pump situated on a 3-m high dam remove water from the lake.

During this study, Cow Head Lake bed was dry except for Cow Head Ditch, water feeding the ditch from Eightmile Creek, and several hectares of marshy habitat just west of Cow Head Ditch. Flow through Cow Head Slough was intermittent with springs

feeding along its course, which was also true of other stream systems sampled in this study (Barrel, West Barrel, Keno, Rock, and Horse creeks). Other similarities among the systems were near-zero gradients in reaches with spring flow, and watershed consisting of high plateau and hills. These physiographic characteristics created

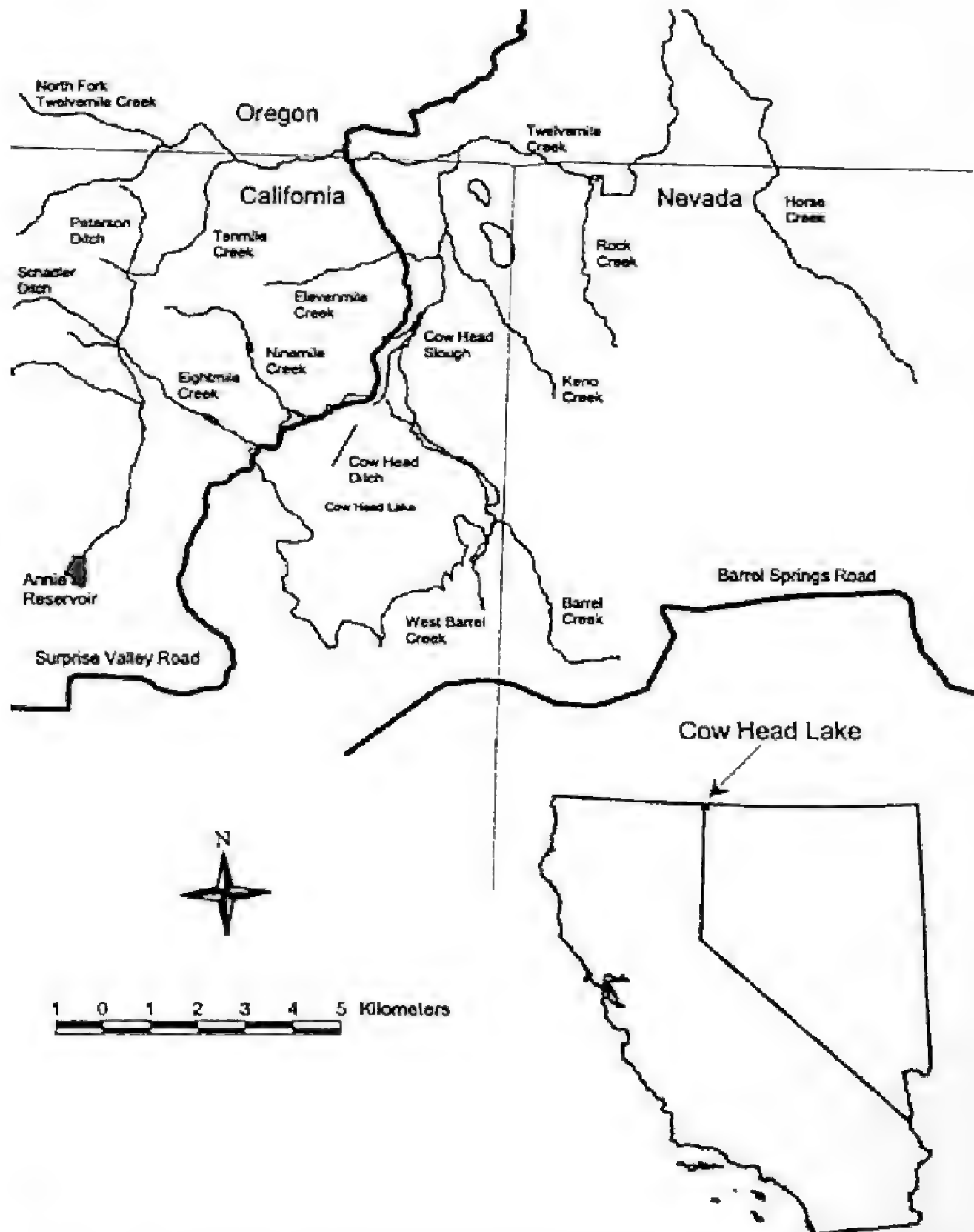


Figure 1. Map of southern Warner Basin showing Cow Head Slough Drainage, Rock Creek, and Horse Creek.

aquatic habitat consisting of spring-fed pools and slack water, except during the short snowmelt runoff period in late winter to early spring.

In recent years, Cowhead Lake tui chub have been captured from Cow Head Slough and Cow Head Ditch (Moyle 2002, Georgina Sato-Lampman, U.S. Forest Service, personal communication). There is no record of fish surveys in the eastside spring systems (Keno, Barrel, and west Barrel) of the Cow Head Basin prior to this survey.

In August and September 2001, we sampled the Cow Head Basin and two spring-fed streams (Rock and Horse creeks) outside the Cow Head Basin. Rock and Horse creeks run parallel and east of the Cow Head Lake Basin and are also tributaries to Twelvemile Creek. We sampled reaches of the drainage where landowners granted permission: Cow Head Ditch on 20 August 2001; Cow Head Slough from Twelvemile Creek upstream to Cow Head Ditch on 21 August 2001; Barrel Creek from Barrel Springs Road to Cow Head Lake on 21 August 2001; West Barrel Creek from its spring origin to Barrel Creek on 29 August 2001; and Keno Creek from its spring origin to Cow Head Slough on 29 August 2001. We sampled Rock Creek 19 and 20 September 2001 and Horse Creek on 1 and 2 October 2001. Rock Creek was surveyed from 1.3 km upstream of Barrel Spring Road to 9.3 km downstream, and Horse Creek was surveyed from 1 km downstream from the California/Oregon border to 5 km upstream of the border.

Our primary method of sampling fish in California and Nevada was electrofishing (Dirigo 750 or Smith-Root VII electrofisher), sampling intermittently along each spring system. In Oregon, where we did not have a permit for sampling fishes, we determined fish presence and identification visually from the banks. In Cow Head Ditch, we used a 9.6-m long, 1.3 m high, 5-mm mesh seine and made one pass at each of four sites approximately 200 m apart. In the 30-m diameter headwater pond of West Barrel Creek we used 14 Gee-type minnow traps baited with dry dog food and placed at regular intervals around the periphery. Traps were fished overnight for about 18 hours.

All captured fish from Cow Head Ditch were measured (fork length-FL) then released. In Cow Head Slough and other streams sampled, up to 25 fish of each species were captured at random and measured. We used the Global Positioning System to mark fish location and later rectified these data on topographic maps so we could illustrate fish distribution using Geographic Information System (GIS).

We found tui chub along the entire length of Cow Head Ditch and along approximately 3 contiguous km out of the 7 km stream of Cow Head Slough (Figure 2). Additional populations were found along 4 km of Barrel Creek, 1 km of West Barrel Creek, and about 0.5 km of Keno Creek (Figure 2). Speckled dace, *Rhinichthys osculus*, was captured from each of the above sites except for Cow Head Ditch. Tui chub were not encountered outside the Cow Head Basin. Redband trout, *Oncorhynchus mykiss* sp., and speckled dace were the only species in Rock Creek and speckled dace the only fish in Horse Creek.

Samples from Cow Head Ditch suggest the local population of tui chub were quite abundant. In four seine hauls, we collected 936 tui chubs, in spite of our seine rolling due to mud and submerged vegetation. Size ranged from 22 mm to 148 mm FL; all fish appeared to be in excellent condition and there was no sign of disease. The largest chub taken was from West Barrel Creek, 190 mm FL (165 mm Standard Length-SL; Figure 3).

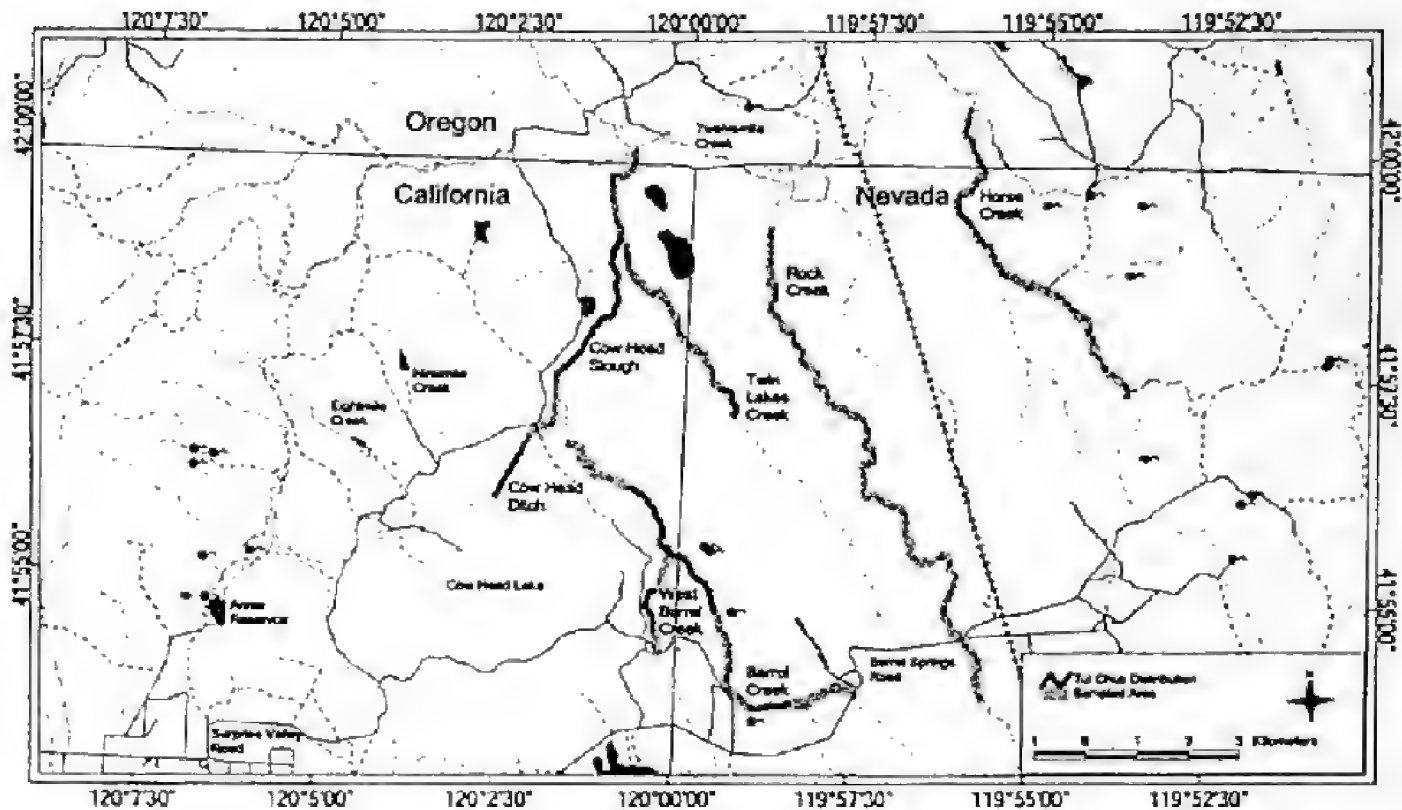


Figure 2. Map of stream reaches surveyed and distribution of Cowhead Lake tui chub.

This survey indicated that tui chub in the Cow Head Basin was more widespread than had been previously reported. Besides the chub known from Cow Head Ditch and Slough, we found additional populations in Barrel Creek, West Barrel Creek, and Keno Creek, all fed by perennial springs. These populations may have been overlooked in the past because tui chub is typically associated with lake, pond, or slow river habitat (Bond et al. 1988, Moyle 2002) rather than with small streams, and so small streams were not sampled. Because these systems are spring-fed in areas of near-zero gradient and runoff of short duration, they were lentic 10 to 11 months each year. Absence of chub from Rock or Horse Creek, also low gradient streams, is open to conjecture. In the case of Rock Creek, the presence of redband trout may have precluded or eliminated tui chub. Furthermore, both systems have a steep gradient as they enter Twelvemile Creek, a probable impediment to the migration of chub (Elias Flores, Bureau of Land Management, personal communication).

Our survey took place during drought, but conditions were not as severe as observed in summer and fall 1992. Had the survey taken place then, chub distribution may have been somewhat less than we observed. For example, there was less than 0.4 km of surface water in Cow Head Slough in summer 1992 (Georgina Sato-Lampman, U.S. Forest Service, personal communication). Surface water level in Barrel, West Barrel, and Keno creeks in summer 1992 is unknown but was probably somewhat lower than observed in this study. However, there are no indications that any of the spring habitats in which tui chub populations were found went dry or that fish populations were lost.

We found a few chub within the Cow Head Basin that were much larger than the

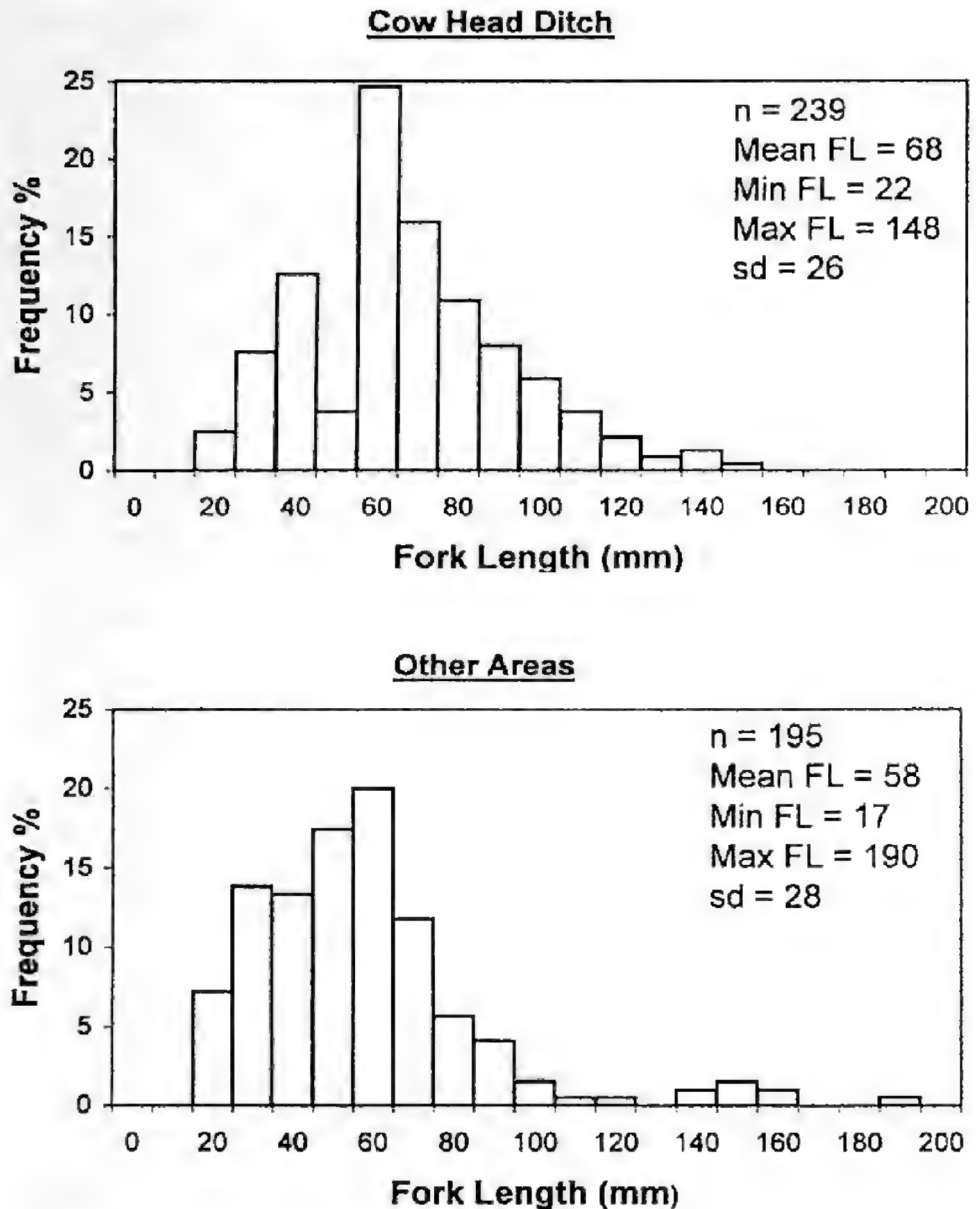


Figure 3. Length frequency of Cowhead Lake tui chub taken from Cow Head Ditch and from other areas; Cow Head Slough, Barrel Creek, West Barrel Creek, and Keno Creek.

maximum length of 116 mm SL described for the subspecies (Bills and Bond 1980). In the summer of 2002, we sampled tui chub in the Cow Head Basin for an age growth study, and a 235 mm FL (\approx 200-mm SL) chub was taken from West Barrel Creek, the largest we captured (unpublished data). Chub in small spring systems such as found in Cow Head Basin would be expected to be shorter than 200 mm (Moyle 2002). Fish tend to scale to their environment (Smith 1981), suggesting that the large fish captured in West Barrel may have moved in from a larger system. At less than a meter deep, the headwater pond on West Barrel is an unlikely source, and was found to contain only a few large speckled dace. We suspect the large fish may have moved in from a larger water body further down in the system. We did not investigate whether these large fish fit the morphometric and meristic description of Cowhead Lake tui chub.

The results of this study indicate that the Cowhead Lake tui chub is more widely distributed in the Cow Head Basin than previously recognized. The newly discovered populations were found in spring-fed stream systems, and characterized by isolated summer pools. Size frequency data suggest similar population structure in all five permanent habitats available to the tui chub, with occasional larger individuals encountered in West Barrel Creek. This improved understanding of Cow Head Basin tui chub distribution was made possible through cooperation of local landowners and will hopefully aid managers and landowners in developing effective stewardship strategies to ensure the long-term survival of the Cowhead Lake tui chub.

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